CHAPTER 10

Aquatic Communities

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Description

Wisconsin has a large and diverse aquatic resource which supports numerous species, communities, ecological processes, and human uses. In addition, many terrestrial species and processes are dependent on neighboring aquatic systems. On a landscape scale, aquatic systems are one integral piece of a larger continuum that includes upland terrestrial systems and transitional wetland areas. The location of a species or community along this continuum is critical to understanding its role in the landscape ecosystem.

Wisconsin waters have been classified based on geographic locations. Frey (1963) identified four major geographic regions: driftless area, northwestern lakes district, northeastern lakes district, and southeastern lakes district. A classification based on nationally identifiable ecoregions was proposed by Omernik and Gallant (1988). Most of Wisconsin lies in four of these ecoregions: northern lakes and forests (NOLF), north-central hardwood forest (NCHF), driftless area (DRFT), and southeastern Wisconsin till plain (SETP) (Fig. 17). Lyons (1989*a*) demonstrated that Wisconsin stream fish communities show a general correspondence with these ecoregions. Other ecoregion classifications have been developed (e.g., Bailey 1989*a*, 1989*b*) and will be used by the Department to develop a classification system for the state.

Most classifications agree that the driftless area is the dominant Wisconsin geologic aquatic boundary. Covering an area missed during the last glaciation, the driftless area is distinguished by classic dendritic stream patterns, few natural lakes, and sharper, more eroded terrain (Becker 1983). In contrast, the remainder of the state was smoothed by glaciation and has less topographic relief. Rivers are sinuous and have less average elevation drop. Glaciers also left substantial numbers of natural lakes. Lakes in northern Wisconsin tend to be cooler, more oligotrophic, and less productive than southern Wisconsin lakes. North-central Wisconsin also has one of the highest concentrations of spring-fed lakes and streams in the world.

Understanding the issues affecting aquatic biological diversity in Wisconsin must involve some generalization of aquatic ecosystem types. A general physical classification includes drainage lakes—impounded or natural lakes whose main water source is from stream drainage and have at least one inlet and one outlet; seepage lakes—landlocked natural lakes whose main water source is the groundwater table and with no inlet or permanent outlet; spring lakes—natural lakes for which the main water source is the groundwater table (springs) but always have an outlet of substantial flow; streams—smaller, low-order flowing waters which form the headwaters of river systems and which usually have a high-moderate gradient; and rivers—larger flowing waters formed by the confluence of several streams and that usually have a low gradient. The lake classifications are derived from the Wisconsin Department of Natural Resources Surface Water Resources program (e.g., Carlson and Andrews 1977).

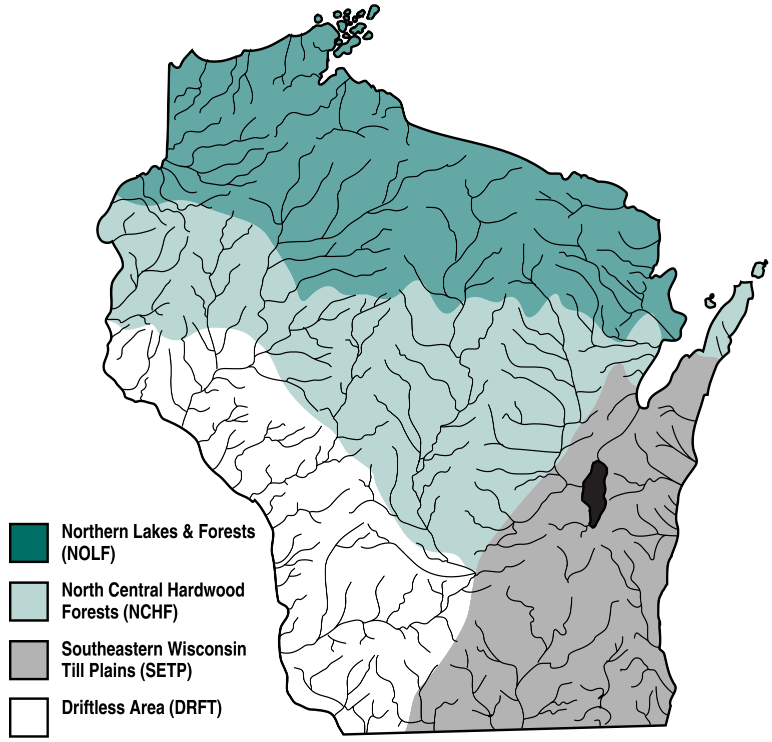

Figure 17

Ecoregions of Wisconsin as developed by Omernik and Gallant (1988). This system is used in this chapter and is one example of how ecoregions could be defined for Wisconsin.

Lakes

Lake communities often vary dramatically based on limnological characteristics. Lakes are often classified according to trophic status. Lakes with very low nutrient input and abundant dissolved oxygen levels throughout the water column are termed "oligotrophic." Oligotrophic lakes, like Lake Superior, are often considered to be the epitome of desirable water quality conditions but have low overall productivity, few species, and relatively simple ecological systems. Conversely, lakes with high nutrient input or high rates of nutrient recycling are termed "eutrophic." Eutrophic lakes that thermally stratify may become devoid of oxygen below the summer thermocline, precluding the production of many species. Eutrophic lakes have high overall productivity and typically support high species diversities and more complex ecological systems. Intermediate lakes with moderate nutrient levels and occasional oxygen depletion are sometimes termed "mesotrophic." Wisconsin also has a special class of lakes termed "dystrophic" or bog lakes, which are primarily affected by natural acidity despite having typical ranges of nutrient input. These dystrophic lakes contain unique communities that have very low species diversity and are among the simplest of ecological systems.

Wisconsin has a large and diverse aquatic resource which supports numerous species, biological communities, ecological processes, and human uses.

Lakes normally undergo a natural succession from oligotrophic to eutrophic although the time span may be thousands of years. Human intervention can shorten this process to a few decades. Lakes receiving unnaturally high nutrient inputs—termed "hyper-eutrophic"—have degraded habitat that results in simplified communities, altered species compositions, and dysfunctional ecological processes.

On a landscape scale, aquatic systems are one integral piece of a larger continuum that includes upland terrestrial systems and transitional wetland areas.

Great Lakes

Wisconsin waters include 1.7 million acres of Lake Superior (Wis. Dep. Nat. Resour. 1988) and 4.7 million acres of Lake Michigan (Wis. Dep. Nat. Resour. 1986) including most of Green Bay. Wisconsin has 156 miles of shoreline along Lake Superior and 407 miles of coastline along Lake

With 6.4 million acres of surface water and 563 miles of shoreline, Wisconsin's Great Lakes represent an immense resource. Geologic features, such as this exposed dolomite along the Lake Michigan shore in Door County, add structural and functional diversity. *Photo by Robert H. Read.*

Michigan (Napoli 1975). Features of national significance include the cobble beach found along only the shoreline of the Door County peninsula; Lake Superior drowned bay mouth estuaries (e.g., St. Louis River, Kakagon Sloughs, and Port Wing) found only along Wisconsin's shore; Lake Michigan drowned bay estuaries (e.g., Marinette, Peshtigo, Green Bay's Atkinson's Marsh) are found primarily along Wisconsin's shoreline; the Apostle Islands National Seashore located in Lake Superior near Ashland; and Lake Superior itself—the second largest freshwater lake in the world.

Understanding the issues affecting aquatic biological diversity in Wisconsin must involve some generalization of aquatic ecosystem types. A general physical classification includes drainage lakes, seepage lakes, spring lakes, streams, and rivers.

The fish communities of Wisconsin's Great Lakes are characteristic of north temperate oligotrophic and mesotrophic lakes.[4] Cold-water communities with lake trout, rainbow trout, brown trout, and coho and chinook salmon as the top predators dominate, but warm-water communities featuring walleye, smallmouth bass, and northern pike exist in littoral and estuarine areas. Cold-water communities contain panfish and non-game species such as deepwater sculpins, bloater, cisco, lake, round whitefish, ninespine stickleback, longnose suckers, rainbow smelt, alewives, and sea lamprey. Warm-water communities contain yellow perch, burbot, white suckers, lake sturgeon, emerald shiners, and carp. Both communities contain a mix of native and introduced species (Downs 1984, 1986). Wisconsin waters of the Great Lakes at one time supported a complex of seven different cisco species, four of which were endemic to the Great Lakes (Becker 1983, Robins et al. 1991).

The macroinvertebrate fauna of Lakes Michigan and Superior is dominated by amphipods (especially *Pontoporeia*), oligochaetes, nematodes, sphaeriids, and chironomids (Cook and Johnson 1974, Dermott 1978, Nalepa 1989). Over 90 taxa of Chironomidae have been collected from southeastern Lake Michigan alone (Winnell and White 1985). A few types of typically lotic water forms such as heptageniids and hydropsychids are common in near shore areas (Barton and Hynes 1978) as well as being present in deeper water (Selgeby 1974). During the mid-1980s the European cladoceran *Bythotrephes cederstroemi* (BC) became established in Lake Huron and quickly spread to the other four Great

Most of Wisconsin's inland lakes are located in northern Wisconsin. Some, such as this lake in Vilas County, are remote and largely undisturbed. *Photo by Michael Mossman.*

[4] Fish and herptile species, for which data are plentiful, are well described in this discussion; other taxa are mentioned throughout the chapter wherever information was made available by contributors.

Lakes (Garton and Berg 1990). Its impact on native zooplankton communities is unknown. It appears inevitable that BC will eventually spread to inland lakes in the Great Lakes region.

Inland Lakes

Wisconsin has more than 14,000 inland lakes covering a million-plus acres (Table 12). Most lakes are located in the northern part of the state. Using the Omernik and Gallant (1988) system of ecoregions, the NOLF ecoregion contains 9,300 lakes covering 455,000 acres, but 85% are glacial or bog lakes of less than ten acres. The NCHF ecoregion contains another 3,200 lakes covering 223,000 acres. In contrast, the DRFT ecoregion, because of its steep topography, contains very few lakes—only 557 covering 68,000 acres. The SETP ecoregion contains only 6% of Wisconsin's lakes but the region includes Lake Winnebago, at 137,708 acres, the state's largest inland lake. The largest concentration of glacier kettle lakes in the world occurs in the Vilas and Oneida county area (Tonn and Magnuson 1982), and a high concentration of spring ponds occurs in the Forest, Langlade and Oneida county area (Carline and Brynildson 1977).

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Most of these lakes are naturally occurring and of glacial origin. However there are 1,550 dams on state waterways which affect water levels on 666,000 acres (65%) of Wisconsin's inland lakes. A series of hydropower reservoirs on the Wisconsin River system dominate central Wisconsin. The largest reservoirs are Petenwell Flowage (23,040 acres), Castle Rock Flowage (13,955 acres), Big Eau Pleine Reservoir (6,830 acres), Lake DuBay (6,700 acres), and Lake Wisconsin (9,000 acres). Large hydropower reservoirs have also been constructed on the Chippewa-Flambeau river system including Lake Wissota (6,300 acres), Lake Chippewa (15,300 acres), and the Turtle-Flambeau Flowage (13,545 acres). The Mississippi River in Wisconsin has a series of navigation dams which have made existing riverine habitat and backwater areas more lacustrine in character. Smaller reservoirs occur on nearly every river and stream system in the state. Dams have also been built on many natural lakes to control water levels.

Fish communities in Wisconsin's lakes are generally typical of warm-water mesotrophic or eutrophic systems. They are dominated by native species, including largemouth bass, black crappie, northern pike, rock bass, and smallmouth bass. Common insectivores include bluegill, yellow perch, pumpkinseed, and johnny darter (Table 13). The most abundant omnivores are bluntnose minnow, golden shiners, white sucker, and common carp.

	Ecoregion				
Lake Type	Driftless Area	N. Central Hardwood Forest	Northern Lakes and Forest	SE WI Till Plains	All
Seepage					
Number	164	1,837	5,966	404	8,371
Total Acres	1,106	28,253	95,864	8,790	134,013
Drainage					
Number	132	922	2,715	255	4,024
Total Acres	27,548	34,375	146,316	10,494	218,733
Impoundment					
Number	261	447	601	239	1,548
Total Acres	39,249	159,974	213,043	253,749	666,015
All					
Number	557	3,206	9,282	898	13,943
Total Acres	67,903	222,602	455,223	273,033	1,018,761

Table 12

Number and area of Wisconsin lakes, by ecoregion as defined by Omernik and Gallant (1988), based on nearest county boundary.

NOLF	Northern Lakes and Forest
NCHF	North Central Hardwood Forest
DRFT	Driftless Area
SETP	Southeast Wisconsin Till Plains

Table 13

Comparison of percent fish species occurrences at stations in Wisconsin lakes. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Trophic Level*** and Species	Percent of Sampled Lake Stations, by Ecoregion				
	NOLF*	NCHF*	DRFT*	SETP*	Average
Top Piscivores					
Largemouth bass	59.5	66.5	95.5	49.8	67.8
Black crappie	17.3	38.3	31.8	27.3	28.7
Northern pike	21.7	20.8	22.7	13.0	19.6
*Rock bass***	*21.4*	*20.8*	*9.1*	*11.4*	*15.7*
*Smallmouth bass***	*12.6*	*16.9*	*18.2*	*6.4*	*13.5*
Insectivores					
Bluegill	66.7	79.2	68.2	72.2	71.5
Yellow perch	72.0	70.7	36.4	56.7	58.9
Pumpkinseed	43.9	53.0	27.3	51.1	43.8
Johnny darter	44.1	38.9	40.9	20.4	36.1
Logperch	11.5	16.6	40.9	10.2	19.8
Spotfin shiner	2.3	7.9	45.5	11.9	16.9
*Iowa darter***	*32.3*	*21.7*	*0.0*	*13.3*	*16.8*
*Blacknose shiner***	*23.3*	*9.0*	*0.0*	*15.2*	*11.9*
Green sunfish	2.1	8.5	9.1	26.9	11.6
*Spottail shiner***	*7.6*	*3.1*	*27.3*	*7.2*	*11.3*
Common shiner	14.7	15.8	4.5	5.6	10.2
Banded killifish	8.2	11.5	0.0	18.6	9.6
Brook silverside	5.3	1.7	9.1	21.3	9.3
*Blackchin shiner***	*15.2*	*12.1*	*0.0*	*9.2*	*9.1*
Black bullhead	8.8	7.3	4.5	13.5	8.5
Mimic shiner	14.8	4.5	0.0	10.4	7.4
Orangespotted sunfish	0.0	0.0	27.3	1.5	7.2
Brown bullhead	3.9	11.3	0.0	7.2	5.6
Emerald shiner	0.2	2.5	4.5	13.2	5.1

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A variety of herptiles inhabit lakes throughout the state (Table 14). Some amphibians use lakes, particularly their shallow bays, for reproduction. In many instances these are marginal breeding habitats with the exception of species dependent on permanent water, such as the bull, green, mink, and Blanchard's cricket frogs. The totally aquatic mudpuppy lives its entire life on the bottom of lakes, usually in deep water (Vogt 1981). All other Wisconsin amphibians rely on ephemeral waters for primary production. Five species of turtles occupy natural lakes including the state-threatened Blanding's turtle. While all five also occupy streams and rivers, all but the eastern spiny softshell are most productive in lake environments. All but the common musk turtle, which is limited to the SETP and DRFT ecoregions, are found in all ecoregions of the state.

Trophic Level*** and Species	Percent of Sampled Lake Stations, by Ecoregion				
	NOLF*	NCHF*	DRFT*	SETP*	Average
Omnivores					
Bluntnose minnow	55.3	55.5	45.5	50.9	51.8
Golden shiner	24.8	21.4	27.3	23.1	24.1
White sucker	25.8	21.4	13.6	11.4	18.0
Common carp	0.2	6.2	31.8	12.2	12.6
Fathead minnow	11.2	15.8	9.1	8.7	11.2
Bullhead minnow	0.0	0.6	13.6	0.2	3.6
Total stations sampled	660	357	22	624	1,644

*NOLF = Northern Lakes and Forest
NCHF = North Central Hardwood Forest
DRFT = Driftless Area
SETP = Southeast Wisconsin Till Plains
**Italics indicate a fish species intolerant of environmental degradation, as defined by Lyons (1992)
***Trophic level as defined by Lyons (1992)

Table 13 (cont'd)

Comparison of percent fish species occurrences at stations in Wisconsin lakes. Includes only fish species found at $\geq$ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Table 14

Herptile species occurring in Wisconsin lakes, classified by ecoregions as defined by Omernik and Gallant (1988).

Species Name	NOLF*	NCHF*	DRFT*	SETP*
Blue-spotted salamander**	▲	▲	▲	▲
Central newt	▲	▲	▲	▲
Eastern tiger salamander**	▲	▲	▲	▲
Mudpuppy	▲	▲	▲	▲
Spotted salamander**	▲	▲		▲
Blanchard's cricket frog[E]			▲	▲[H]
Bullfrog[SC]	▲	▲	▲	▲
Cope's gray treefrog**	▲	▲	▲	▲
Eastern American toad**	▲	▲	▲	▲
Eastern gray treefrog**	▲	▲	▲	▲
Green frog	▲	▲	▲	▲
Mink frog	▲	▲		
Northern leopard frog**	▲	▲	▲	▲
Spring peeper**	▲	▲	▲	▲
Western chorus frog**	▲	▲	▲	▲
Blanding's turtle[T]	▲		▲	▲
Common snapping turtle	▲	▲	▲	▲
Common Map turtle	▲		▲	▲
Common musk turtle			▲	▲
Eastern spiny softshell turtle	▲	▲	▲	▲
Western/Midland painted turtle	▲	▲	▲	▲
Northern water snake	▲	▲	▲	▲

*NOLF = Northern Lakes and Forest
NCHF = North Central Hardwood Forest
DRFT = Driftless Area
SETP = Southeast Wisconsin Till Plains
** = Breeding Habitat Only
E = State Endangered
T = State Threatened
SC = Special Concern
H = Historic

The macroinvertebrates of Wisconsin's inland lakes have not been intensively studied at a statewide scale. Preliminary indications suggest that species of Chironomidae would make up 75% or more of the taxa for most lakes (Richard Narf, Wis. Dep. Nat. Resour., pers. comm.). Most species of Hemiptera, Coleoptera, and Diptera occur solely or predominantly in inland lakes. There are no federal or state endangered or threatened aquatic insects for which inland lakes form primary habitat.

Rivers and Streams

Wisconsin's rivers and streams do not form distinct trophic states. Energy systems and species assemblages typically form a continuum from smaller, upstream headwaters to larger, downstream rivers (Vannote et al. 1980, Minshall et al. 1985). Rivers and streams may be classified into orders according to the number of branches or divisions from their mouth to their source (Strahler 1957). Lyons et al. (1988) showed that there is considerable gradation of fish

A diverse system of headwater streams and tributaries feed into larger streams and rivers throughout the state. Shown here is Mecan River in central Wisconsin, which supports a trout fishery and diverse macroinvertebrate community. *Photo by Staber Reese.*

species along Wisconsin's flowing water habitats. Rivers and streams may also be classified by water temperature into warm-water, cool-water, and cold-water systems. Species inhabiting these systems usually reflect the maximum tolerable temperature limiting the presence of various aquatic species.

In Wisconsin, rivers and streams are commonly classified by fish community types. Smaller, spring-fed headwater streams and some rivers in the northern part of the state can support a fish community with trout or salmon as the top fish predator. Smaller streams fed by surface water or located in the southern part of the state are typically warmer and support fish communities with smallmouth bass as the top fish predator. Larger rivers support only warm-water fish communities with smallmouth bass, walleye, largemouth bass, northern pike, or muskellunge as the top fish predators. Rivers and streams with trout or salmon are often classed as "cold-water" systems, while the other streams and rivers are often classed as "warm-water" systems. Cold-water systems are afforded special protection under state law.

Streams

This category includes rivers and streams with mean annual flows of 40 cms or less (Lyons 1992). A definitive inventory of Wisconsin's streams is not available, but Becker (1983) indicates that of the 33,000 miles of rivers and streams in the state, 9,561 miles are cold-water trout streams (Wis. Dep. Nat. Resour. 1980). Adequate natural trout reproduction occurs in only 37% of the state's cold-water streams. The status of warm-water fish populations on most warm-water streams is not well known.

Table 15

Comparison of percent fish species occurrences at stations in Wisconsin streams. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Trophic Level*** and Species	Percent of Sampled Stream Stations, by Ecoregion				
	NOLF*	NCHF*	DRFT*	SETP*	Average
Top Piscivores					
*Brook trout***	*46.5*	*32.0*	*15.9*	*3.4*	*24.5*
Northern pike	11.5	18.9	7.6	28.7	16.7
Brown trout	13.4	21.4	22.0	7.9	16.2
*Rock bass***	*11.3*	*18.7*	*3.7*	*14.0*	*11.9*
Largemouth bass	6.8	12.3	6.8	18.7	11.2
*Smallmouth bass***	*4.3*	*13.6*	*11.5*	*10.8*	*10.0*
Burbot	14.0	10.8	3.3	0.5	7.1
Black crappie	2.5	7.1	2.8	11.4	6.0

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Table 15 (cont'd)

Comparison of percent fish species occurrences at stations in Wisconsin streams. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Trophic Level*** and Species	Percent of Sampled Stream Stations, by Ecoregion				
	NOLF*	NCHF*	DRFT*	SETP*	Average
Insectivores					
Creek chub	62.1	57.6	62.4	49.1	57.8
Johnny darter	33.7	50.1	65.3	40.3	47.4
Common shiner	44.2	48.9	29.1	38.7	40.2
Central mudminnow	48.3	52.4	16.8	40.5	39.5
Brook stickleback	40.4	31.4	39.0	32.8	35.9
Blacknose dace	45.0	38.6	40.3	15.8	34.9
*Mottled sculpin***	*45.1*	*29.7*	*7.1*	*10.6*	*23.1*
Hornyhead chub	22.0	26.2	19.0	18.4	21.4
Fantail darter	5.4	19.1	30.4	14.6	17.4
Longnose dace	20.4	21.6	22.8	4.5	17.3
Black bullhead	6.8	18.8	5.2	35.5	16.6
Pearl dace	25.2	25.5	2.0	6.5	14.8
Blackside darter	9.3	26.6	10.8	10.1	14.2
Green sunfish	0.3	6.0	10.7	39.4	14.1
Pumpkinseed	6.2	19.7	4.1	21.4	12.9
Bigmouth shiner	0.8	8.8	26.0	12.9	12.1
Spotfin shiner	0.9	7.9	21.1	18.5	12.1
Bluegill	6.8	12.6	6.7	21.4	11.9
*Northern hog sucker***	*8.3*	*22.1*	*10.5*	*6.5*	*11.8*
Yellow perch	15.0	12.7	1.6	13.0	10.6
Shorthead redhorse	5.6	7.1	14.8	8.9	9.1
Yellow bullhead	2.8	6.6	2.3	18.2	7.5
Stonecat	1.0	7.9	8.7	11.4	7.2
Sand shiner	1.1	2.7	9.5	14.5	6.9
*Rosyface shiner***	*1.1*	*10.4*	*9.0*	*4.0*	*6.1*
*Blacknose shiner***	*10.3*	*10.2*	*0.4*	*2.9*	*5.9*
*Banded darter***	*0.4*	*10.9*	*6.2*	*5.8*	*5.8*
*Rainbow darter***	*0.8*	*11.9*	*1.9*	*4.8*	*4.9*
Finescale dace	12.5	2.3	0.0	0.1	3.7
Suckermouth minnow	0.0	0.1	10.2	3.9	3.5

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A wide variety of warm- and cold-water fish species are found in Wisconsin streams (Table 15). Common species include brook trout, creek chub, johnny darter, common shiner, central mudminnow, brook stickleback, blacknose dace, white sucker, bluntnose minnow, and fathead minnow.

Knowledge about macroinvertebrates of Wisconsin's streams is still at the descriptive stage where distributions of species are becoming reasonably well known for many

Table 15 (cont'd)

Comparison of percent fish species occurrences at stations in Wisconsin streams. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Trophic Level[***] and Species	Percent of Sampled Stream Stations, by Ecoregion				
	NOLF[*]	NCHF[*]	DRFT[*]	SETP*	Average
Omnivores					
White sucker	60.8	70.3	70.3	70.8	68.0
Bluntnose minnow	12.0	29.6	42.1	42.2	31.5
Fathead minnow	16.0	30.2	30.3	40.4	29.2
Common carp	0.2	6.4	10.7	31.8	12.3
Golden shiner	7.3	10.6	4.0	13.4	8.8
Herbivores					
Central stoneroller	0.2	6.9	33.4	15.4	13.9
Brassy minnow	20.6	15.9	9.2	8.4	13.5
Northern redbelly dace	27.7	18.3	0.8	5.0	12.9
Southern redbelly Dace	0.0	1.9	20.3	13.7	9.0
American brook lamprey[**]	*2.6*	*7.0*	*16.8*	*2.2*	*7.2*
Largescale stoneroller	5.9	15.4	2.5	3.8	6.9
Total stations sampled	1,317	1,079	1,586	1,433	5,415

[*]NOLF = Northern Lakes and Forest
NCHF = North Central Hardwood Forest
DRFT = Driftless Area
SETP = Southeast Wisconsin Till Plains

[**] Italics indicate a species intolerant of environmental degradation, as defined by Lyons (1992)
[***] Trophic levels as defined by Lyons (1992)

orders but significant gaps in knowledge remain. Overall, the number of streams that have been studied in detail is small. No effort has been made to compare macroinvertebrate faunas among ecoregions. Aquatic arthropods can be used to evaluate the water quality of streams based on the tolerance of the taxa to organic and nutrient pollution (Hilsenhoff 1987). Most species of Plecoptera, Ephemeroptera, and Trichoptera are found solely or predominantly in streams. Two dragonflies, two mayflies, and one riffle beetle that inhabit streams and rivers are listed as state-endangered. Additionally, three dragonflies are listed as state-historical, suggesting they have been extirpated from Wisconsin waters.

Three state listed species of stream freshwater mussels, ellipse, rainbow shell, and slippershell, were once widespread in the DRFT and SETP ecoregions. Geographic ranges have decreased over 90% for these species. They are riffle species preferring clear, small, warm-water streams and have been negatively affected by sedimentation, dam construction, fish community manipulations, and point pollution discharges. They are now restricted to small reaches in watersheds where these effects have been minimal. The rainbow shell remains only in one five-mile reach of one of the most well preserved SETP streams and is in immediate danger of extirpation from effects of urban sprawl.

Several herptile species occupy streams in Wisconsin (Table 16). The queen snake exclusively inhabits streams and their riparian corridors in the SETP ecoregion. This state-endangered snake, while on the northern fringe of its range, has declined in recent history as a result of water quality degradation including sedimentation and turbidity. The specific microhabitat of this species in the stream, flat rocky substrate, has been inundated by sediments throughout much of its former range in southeastern Wisconsin. The Blanchard's cricket frog, dependent on stream habitat and Wisconsin's most endangered herptile, has seen a marked reduction in its range in Wisconsin and elsewhere throughout the northern limits of its distribution.

Large Rivers

Large rivers are those having a mean annual flow of 40 cms or larger (Lyons 1992). Wisconsin has 11 stretches of large rivers: the Mississippi River, the Wisconsin River below Tomahawk, the Chippewa River below the mouth of the Flambeau River, the St. Croix River below the mouth of the Clam River, the Fox River below the mouth of the Puchyan River and between Lake Winnebago and Green Bay, the Menominee River below the Highway 2/141 bridge, the Rock River below Lake Koshkonong, the Flambeau River below the confluence of the north and south forks, the Wolf River below Shiocton, the Black River in LaCrosse County, and the Red Cedar River below Menomonie. Most of these river stretches have been dammed to produce hydropower.

These large rivers support only warm-water fish communities. The most abundant large predators are northern pike, walleye, smallmouth bass, largemouth bass, channel catfish, and burbot (Table 17). Common middle trophic level species are bluegill, black crappie, yellow perch, rock bass, pumpkinseed, freshwater drum, and white bass. A large number of lower trophic level species have been found at sampled river stations, but the most common are spotfin shiner, shorthead redhorse, golden redhorse, sand shiner, emerald shiner, common carp, johnny darter, logperch, northern hog sucker, white sucker, silver redhorse, and bluntnose minnow.

Wisconsin's large rivers contain some of the highest freshwater mussel species richness remaining in North America. The Wisconsin River contains 42 taxa, and the St. Croix has 39. Some southern United States rivers contained more species but

The Mississippi River and Wisconsin River, shown here at their confluence, are the most dominant riverine features in the state. They are biologically rich and provide a major corridor for movement of species throughout the watershed and region. *Photo by Ken Beghin*

Table 16

Herptile species occurring in Wisconsin streams, classified by ecoregions as defined by Omernik and Gallant (1988).

Species Name	NOLF*	NCHF*	DRFT*	SETP*
Four-toed salamander**	▲	▲	▲	▲
Mudpuppy	▲	▲	▲	▲
Blanchard's cricket frog[E]			▲	▲[H]
Bullfrog[SC]	▲	▲	▲	▲
Green frog	▲	▲	▲	▲
Mink frog	▲	▲		
Pickerel frogs	▲	▲	▲	▲
Blanding's turtle[T]	▲		▲	▲
Common snapping turtle	▲	▲	▲	▲
Common musk turtle[M]			▲	▲
Eastern spiny softshell turtle	▲	▲	▲	▲
Western/Midland painted turtle[M]	▲	▲	▲	▲
Wood turtle	▲	▲		
Northern water snake	▲	▲	▲	▲
Queen snake[E]				▲

*NOLF = Northern Lakes and Forest
NCHF = North Central Hardwood Forest
DRFT = Driftless Area
SETP = Southeast Wisconsin Till Plains

** = Breeding Habitat Only
E = State Endangered
T = State Threatened
SC = Special Concern
M = Marginal Habitat
H = Historic

Table 17

Comparison of percent fish species occurrences at Wisconsin river stations. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Trophic Level*** and Species	Percent of Sampled Stream Stations, by Ecoregion				
	NOLF*	NCHF*	DRFT*	SETP*	Average
Top Piscivores					
Northern pike	54.4	36.0	23.0	38.5	38.0
Walleye	45.6	26.7	31.7	46.2	37.5
*Smallmouth bass***	*39.7*	*37.3*	*28.1*	*42.3*	*36.9*
Black crappie	22.1	21.3	42.0	34.6	30.0
*Rock bass***	*30.9*	*24.0*	*29.9*	*7.7*	*23.1*
Largemouth bass	4.4	13.3	42.9	19.2	20.0
Channel catfish	17.6	12.0	11.5	23.1	16.1
White bass	0.0	4.0	31.4	15.4	12.7
Burbot	32.4	9.3	0.6	3.8	11.5
White crappie	1.5	1.3	19.3	23.1	11.3
Sauger	0.0	5.3	22.4	11.5	9.8
Bowfin	2.9	2.7	11.2	7.7	6.1
Longnose gar	0.0	2.7	21.8	0.0	6.1
Yellow bass	0.0	0.0	6.6	15.4	5.5
Shortnose gar	0.0	4.0	10.9	0.0	3.7
Insectivores					
Spotfin shiner	51.5	70.7	67.4	50.0	59.9
Shorthead redhorse	52.9	44.0	37.8	42.3	44.3
Bluegill	16.2	37.3	58.6	42.3	38.6
Golden redhorse	63.2	40.0	16.3	15.4	33.7
Sand shiner	29.4	42.7	27.8	34.6	33.6
Emerald shiner	0.0	38.7	66.8	26.9	33.1
Johnny darter	29.4	32.0	41.4	15.4	29.5
Logperch	30.9	32.0	30.5	23.1	29.1
*Northern hog sucker***	*54.4*	*33.3*	*5.4*	*23.1*	*29.1*
Silver redhorse	52.9	34.7	13.0	11.5	28.0
Yellow perch	39.7	24.0	38.1	7.7	27.4
Common shiner	69.1	21.3	3.6	0.0	23.5
Mimic shiner	27.9	38.7	12.4	0.0	19.7
River shiner	0.0	6.7	53.2	7.7	16.9
Brook silverside	7.4	17.3	31.1	7.7	15.9
Pumpkinseed	2.9	6.7	29.3	23.1	15.5
*Spottail shiner***	*2.9*	*2.7*	*43.8*	*11.5*	*15.2*
Freshwater drum	0.0	8.0	33.8	19.2	15.3
River redhorse	29.4	24.0	1.5	0.0	13.7
Western sand darter	0.0	17.3	20.2	11.5	12.3

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Trophic Level*** and Species	Percent of Sampled Stream Stations, by Ecoregion NOLF*	NCHF*	DRFT*	SETP*	Average
Hornyhead chub	41.2	2.7	0.9	0.0	11.2
Blackside darter	29.4	12.0	2.7	0.0	11.0
*Greater redhorse***	*25.0*	*10.7*	*0.0*	*3.8*	*9.9*
*Gilt darter***	*32.4*	*6.7*	*0.0*	*0.0*	*9.8*
Mooneye	0.0	14.7	16.3	7.7	9.7
Pugnose minnow	0.0	6.7	27.8	3.8	9.6
River darter	0.0	16.0	13.0	7.7	9.2
Smallmouth buffalo	0.0	8.0	13.3	15.4	9.2
Green sunfish	0.0	2.7	2.4	30.8	9.0
*Spotted sucker***	*0.0*	*8.0*	*23.3*	*3.8*	*8.8*
Bigmouth buffalo	0.0	4.0	6.3	23.1	8.4
*Blue sucker***	*1.5*	*9.3*	*7.6*	*11.5*	*7.5*
Black bullhead	4.4	4.0	0.9	19.2	7.1
Yellow bullhead	1.5	2.7	8.8	15.4	7.1
Bigmouth shiner	1.5	2.7	11.5	11.5	6.8
*Slenderhead darter***	*10.3*	*5.3*	*3.3*	*7.7*	*6.7*
Tadpole madtom	1.5	1.3	18.1	3.8	6.2
*Speckled chub***	*0.0*	*2.7*	*10.0*	*11.5*	*6.0*
Stonecat	5.9	0.0	1.8	15.4	5.8
*Banded darter***	*0.0*	*6.7*	*3.0*	*11.5*	*5.3*
Silver chub	0.0	0.0	17.8	0.0	4.5
Orangespotted sunfish	0.0	0.0	17.5	0.0	4.4
Paddlefish	0.0	2.7	1.2	11.5	3.9
Creek chub	10.3	2.7	1.8	0.0	3.7
Longnose dace	14.7	0.0	0.0	0.0	3.7
Mud darter	0.0	0.0	10.3	0.0	2.6
Omnivores					
Common carp	8.8	33.3	40.5	46.2	32.2
White sucker	32.4	36.0	12.7	34.6	28.9
Bluntnose minnow	23.5	37.3	16.6	26.9	26.1
Quillback	8.8	25.3	37.8	23.1	23.7
Bullhead minnow	0.0	4.06	2.2	7.7	18.5
Golden shiner	7.4	9.3	29.0	3.8	12.4
*Highfin carpsucker***	*0.0*	*13.3*	*8.5*	*15.4*	*9.3*
Gizzard shad	0.0	0.0	28.4	7.7	9.0
Fathead minnow	7.4	5.3	4.8	11.5	7.3
River carpsucker	0.0	2.7	10.9	7.7	5.3

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Table 17 (cont'd)

Comparison of percent fish species occurrences at Wisconsin river stations. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

Table 17 (cont'd)

Comparison of percent fish species occurrences at Wisconsin river stations. Includes only fish species found at ≥ 10% of stations in at least one region, as defined by Omernik and Gallant (1988).

	Percent of Sampled Stream Stations, by Ecoregion				
Trophic Level* and Species**	**NOLF***	**NCHF***	**DRFT***	**SETP***	**Average**
Herbivores					
Brassy minnow	17.6	8.0	5.7	3.8	8.8
*Mississippi silvery minnow***	*0.0*	*2.7*	*18.1*	*0.0*	*5.2*
Parasites					
Chestnut lamprey	48.5	14.7	3.9	3.8	17.7
Silver lamprey	1.5	10.7	3.6	7.7	5.9
Total stations sampled	68	75	331	26	500

*NOLF = Northern Lakes and Forest
NCHF = North Central Hardwood Forest
DRFT = Driftless Area
SETP = Southeast Wisconsin Till Plains

** Italics indicate a species intolerant of environmental degradation, as defined by Lyons (1992)
*** Trophic levels as defined by Lyons (1992)

many of these species have been eliminated. Many of Wisconsin's listed mussel species have been eliminated or reduced by water level manipulations, commercial harvest, chemical treatments, fish community manipulations, competition from exotics, channelization, dam construction, and point and nonpoint-source pollution.

A variety of herptiles are found in Wisconsin's rivers (Table 18) including the endangered Blanchard's cricket frog found in rivers in the DRFT and the threatened wood turtle found in rivers in the DRFT and NOLF ecoregions. Some of the larger rivers have endangered species of dragonflies. At times, these dragonfly species are limited to specific river reaches. Thus, they are vulnerable to changes in habitat from riprapping, dredging, and modifications of velocities due to bridge construction.

Wisconsin's large rivers contain some of the most diverse freshwater mussel species associations remaining in North America. The Wisconsin River contains 42 taxa, and the St. Croix has 39.

A number of state and federally listed plants are aquatic or riparian, and are associated with river ecosystems. Wisconsin lists ten endangered, ten threatened and 36 species of special concern that are supported by river ecosystems.

A number of state and federally listed plants are aquatic or riparian, and are associated with river ecosystems. Wisconsin lists ten endangered, ten threatened and 36 species of special concern that are supported by river ecosystems.

Past Status

Wisconsin's aquatic communities were shaped by the last glaciation. About 11,000 years ago, ice covered most of what is now Wisconsin, precluding the existence of aquatic communities (Bailey and Smith 1981). The cold and turbid glacial meltwaters draining through the DRFT would have eliminated all but the simplest cold-water communities. As the glaciers retreated, aquatic organisms recolonized Wisconsin's waters from the Bering (Lake Agassiz), upper Mississippi, and Atlantic refugia (Bailey and Smith 1981, Greene 1935, Stewart and Lindsey 1983). The glaciers receded and crustal rebound alternately opened and closed connections between drainages until about 6,000 years ago, when the current physical aquatic landscape emerged.

Quantitative surveys of Wisconsin's aquatic resources were not made until the early 1900s. Consequently, descriptions of Wisconsin's earlier aquatic communities must be deduced from knowledge of current aquatic community status; the few early, usually anecdotal, descriptions of aquatic resources in the state; the few existing paleological studies of aquatic organisms; information on the nature and scope of human activities that have occurred in the state; and our understanding of the impacts such activities can have on aquatic systems.

> Like terrestrial systems, aquatic systems are subject to the effects of simplification and fragmentation. Most major simplification in Wisconsin has been caused by human activity, but natural phenomena such as drought and forest fires have temporarily simplified aquatic systems.

The aquatic resources of the state have been impacted and changed to varying degrees by human activities since the area was repopulated after the last glaciation. Major changes began in the period of logging and rapid agricultural development in the late 1800s and early 1900s and continued through the industrialization of the 1920s to the 1960s into the current residential and recreational development period.

Aquatic systems are subject to simplification and fragmentation impacts just as with terrestrial systems. Most major simplification impacts in Wisconsin have been caused by human activity, but natural phenomena such as drought and forest fires have temporarily simplified aquatic systems. The impacts of simplification have included extirpation of native species, reduced species richness, loss of top predator species, shifts toward more generalized-feeding or more disturbance-tolerant species, reduced community abundance, reduced genetic diversity, and community instability. Such impacts have commonly been caused by direct loss or degradation of habitat, but they have also resulted from more subtle causes such as well-intended management activities (like stocking or chemical treatment), invasions of exotic species, and commercial or sport fishing. Scientists are just beginning to understand the critical importance of flood events and subsequent aquatic-terrestrial interactions in floodplains in shaping the biota of major rivers (Junk et al. 1989). Channel and flow modifications have resulted in simplification of these natural processes.

Fragmentation of aquatic communities is obvious in cases such as dam construction, where migrations of fish or other organisms are blocked. In other cases, severe simplification such as channelization, dredging, or areas of poor water quality have effectively fragmented aquatic communities. Fragmentation isolates populations, thereby increasing the long-term probability of loss of genetic diversity or

Table 18

Herptile species occurring in Wisconsin rivers, classified by ecoregions as defined by Omernik and Gallant (1988).

Species Name	NOLF*	NCHF*	DRFT*	SETP*
Mudpuppy	▲	▲	▲	▲
Blanchard's cricket frog[E]			▲	▲[H]
Bullfrog[SC]	▲	▲	▲	▲
Green frog	▲	▲	▲	▲
Mink frog	▲	▲		
Pickerel frogs[M]	▲	▲	▲	▲
Blanding's turtle[T]	▲		▲	▲
Common map turtle	▲		▲	▲
Common musk turtle			▲	▲
Common snapping turtle	▲	▲	▲	▲
Eastern spiny softshell turtle	▲	▲	▲	▲
False map turtle	▲		▲	
Smooth softshell turtle			▲	
Western/Midland painted turtle	▲	▲	▲	▲
Wood turtle	▲	▲		
Northern water snake	▲	▲	▲	▲
Queen snake				▲

*NOLF = Northern Lakes and Forest
NCHF = North Central Hardwood Forest
DRFT = Driftless Area
SETP = Southeast Wisconsin Till Plains

E = State Endangered
T = State Threatened
SC = Special Concern
M = Marginal Habitat
H = Historic.

Past and Present Actions Causing Concern

- Dam Construction
- Point-Source Pollution
- Agriculture
- Non-Agricultural Nonpoint Source Pollution
- Timber Harvest
- Channelization and Clearing of Streams
- Invasion of Exotic Species
- Riparian Development
- Fish Stocking and Poor Understanding of Genetic Diversity
- Large-Scale Chemical Treatment
- Department Management Priorities
- Habitat Improvement Projects
- Water Level Manipulations
- Estuary Habitat Management
- Lack of Monitoring
- Bioengineering
- Recreation

extinction due to random events. Fragmentation has isolated migratory species from necessary spawning, nursery, or adult habitat. Fragmentation has also interfered with recolonization of aquatic communities suffering from simplification impacts, even after the impacts are corrected.

Past and Present Actions Causing Concern

Dam Construction

Over 3,700 dams of varying sizes have been built on Wisconsin's rivers and streams. During the logging period, permanent and temporary dams were constructed to provide power for saw mills and increased water flow to float logs downstream. These dams were built on almost all major Wisconsin rivers, including the Chippewa, Flambeau, Black, Wisconsin, Peshtigo, Menominee, Oconto, and Iron rivers, and on numerous smaller streams. In the southern part of the state, dams were constructed to operate grain mills or for navigation.

In later years, many of the larger dams were converted to hydroelectric generation to supply power for the paper mills that grew up along the rivers or to generate electricity for residential or industrial use (Stark 1988). Smaller dams were maintained or constructed to create reservoirs and associated lakefront property, control water levels in natural lakes, or control floods. Water level control structures were built in low-lying areas such as Horicon Marsh to create and maintain wetlands for waterfowl habitat. A series of large dams and reservoirs was constructed on the Mississippi River to maintain a navigation channel for barges.

Dams have allowed humans to harness the power of water and have provided recreational benefits in the form of reservoirs. However, dams can simplify and fragment river habitats in a number of ways. *Photo by F. Albert.*

Dam construction can simplify and fragment river habitats in a number of ways. Most obviously, dams change riverine (lotic) habitat into lake or reservoir (lentic or lacustrine) habitat. Since dams are generally built in areas where rivers have a steeper vertical drop, higher gradient riffles and rapids are eliminated. Reservoirs created by dams can increase water temperatures and reduce dissolved oxygen levels in water discharged below the dam. Dramatic changes in stream flow patterns can disrupt spawning of native fish, reduce macroinvertebrate habitat, and increase erosion (Tyus 1990). Meffe (1991) and Winston et al. (1991) showed losses of native species in a river system after impoundments were built. Martinez et al. (1994) documented that even small-scale impoundments that do not radically alter hydrologic or thermal regimes can still have a strong negative influence on native fish by facilitating establishment and proliferation of non-native species.

Dams also interfere with the natural flooding and sediment transport patterns in a river. Natural flooding and sediment flow patterns include periods of scouring and sediment deposition that maintain the complex gravel riffle, pool, run river habitats, and seasonally provide rich nutrients to floodplain areas. Disruptions of these patterns can result in loss of riffle and pool habitat, depletion of nutrients in floodplain areas, and loss of sandbars. Sedimentation in upper reaches of reservoirs can greatly alter wetland areas. Dams interfere with the natural downstream transport of woody debris which forms important habitat for macroinvertebrates, fish, and other aquatic organisms. Logs, brush, and other debris that naturally enter river systems from riparian sources accumulate behind dams leaving downstream areas without this habitat.

Dams are typically impassable to upstream migration and pose mortality threats to downstream-migrating species. The few fish ladders which do exist are old and largely ineffective. No Wisconsin dams are equipped for downstream fish passage so migrating fish are exposed directly to mortality in turbines or spillways.

Dams alter contaminant dynamics within aquatic systems. Spring high flows flush contaminated water and sediments from basins. Blockage of this cleansing can cause accumulation within the reservoir particularly at the dam base. Contaminants in the collected sediments are then available for resuspension in the water column or uptake by bottom-feeding species. The upstream flooding of riverine wetlands produces elevated methyl-mercury in mercury contaminated systems (Zillioux et al. 1993).

In Wisconsin, dam construction and operation has had major impacts on fish. Becker (1983) noted that the gilt darter has been affected by dams because its preferred habitat, which is the large, fast-flowing sections of rivers, has often been used as dam sites. Eddy and Underhill (1974) regarded the gilt darter population in the Saint Croix River as a "modern relict population which has been isolated in recent times by habitat modifications in its former range." The river redhorse, a state-threatened species, is declining in much of its range due to dam construction (Becker 1983).

Fish such as the paddlefish, lake and shovelnose sturgeon, blue sucker, and skipjack herring and several mussel species dependent on these fish for glocidial hosts are examples of species whose range has been dramatically altered by dams (Becker 1983). According to Helms (1974), populations of shovelnose sturgeon have been reduced in the Mississippi River due to habitat destruction resulting from several improvements to the navigation dams and channel civil works. Now shovelnose sturgeon are restricted to areas immediately below navigation dams. Construction of the Keokuk Dam on the Mississippi River (Lock and Dam 19 near Keokuk, Iowa) presented a barrier to extensive upstream

Applying the Ecosystem Management Decision Model to Aquatic Communities

The list of past and present actions causing concern for aquatic communities is lengthy, and the items on the list are complex and interrelated. All together, they point to the many dimensions of the human relationship to water. It is a resource that connects us in a myriad of seen and unseen ways to the components of the ecosystems upon which we depend. How will we make decisions that recognize the role of humans as part of aquatic ecosystems and at the same time fully protect them for future generations?

One positive step we can take is to begin to use and refine the ecosystem management decision model described in the second chapter. This model provides a series of questions that managers can ask to approach decision-making from three perspectives: the ecological, socio-economic, and institutional. Our success as resource stewards is a function of our ability to understand, analyze, and integrate alternatives across all three. The conservation of biological diversity is one of the threads that weaves throughout the model as it is applied to the array of actions that humans take to affect aquatic communities.

The questions and considerations for managers to use to address each of the three contexts are listed in the second chapter. However, there are two that deserve highlighting here. First, it is important that we apply the model on the landscape scale so that recommendations are made using the appropriate geographic boundaries. This will help us ask and answer the kind of broad regional questions that will guide the management of individual lakes. For example, how many lakes of different types are present in a region; what is their past, present, and potential future condition; and what strategies are needed to conserve biological diversity and provide for the range of human uses?

Second, it is clear that DNR is not alone in this work; success will be measured by our ability to identify and include stakeholders and to foster innovative partnerships with other agencies, local governments, and private interests.

migration of paddlefish, American eel, skipjack, Ohio shad, buffalo, shortnose gar, freshwater drum, carp, shovelnose sturgeon, and three species of catfish (Carlander 1954). The dam interfered with sauger movement during the winter, and spawning areas were cut off for the skipjack herring, the Ohio shad, and the blue sucker. The skipjack herring is the glocidial host for the ebony shell and elephant ear mussels. When the herring was extirpated from Wisconsin by construction of the Keokuk dam, the ebony shell and elephant ear mussels became endangered in Wisconsin occurring now only as scattered, old-age individuals (Becker 1983).

Becker (1983) reported that the paddlefish has also been affected by the construction of dams and flood control projects that flood its spawning areas. It was once abundant in Lake Pepin, where its numbers are now considerably reduced. Lyons (1993) noted that paddlefish could not recolonize areas above the Prairie du Sac dam on the Wisconsin River following water quality improvements because the dam prevented upstream movement. Heath (1993*a*) found that at least five mussel species were prevented from upstream recolonization through the same dam.

Becker (1983) made similar observations about the lake sturgeon. He noted hydroelectric dams act as barriers to movement of lake sturgeon, isolating their populations. Since lake sturgeon are long-lived but reproduce slowly, they may persist in an area for a long time, but they are susceptible to pollution, angler exploitation, poaching, and natural morality. Thus they may gradually die out without a source of adequate natural reproduction. High spring flows through the gated section of the dams tend to attract spawning lake sturgeon, inducing some to drop their eggs. Flows through the gates may later be shut, trapping the larger lake sturgeon behind boulders, in plunge pools, and behind riffles (Joseph Kurz, Wis. Dep. Nat. Resour., pers. comm.). Any eggs that were deposited are then exposed to the air and eventual desiccation. The adults are subject to eventual death due to exposure or poaching. Lake sturgeon have also been killed by hydroelectric equipment (Tom Thuemler, Wis. Dep. Nat. Resour., pers. comm.) and found entrained on dam trash racks (Tim Larson, Wis. Dep. Nat. Resour., pers. comm.).

Dams have had an even more dramatic impact on Wisconsin freshwater mussel populations. Mussels often congregate immediately below dams. Dams act as barriers to upstream fish movement and fish are more likely to drop the mussels' parasitic glocidial stage in areas immediately below the dams (Robert Martini, Wis. Dep. Nat. Resour., pers. comm.). The increased velocities through the reach below the dams may help scour the mussel beds clean of sediments. The upstream reservoirs probably also help to supply algae, diatoms, and other microscopic organisms that are food for filter feeders such as mussels (e.g., Ney and Mauney 1981), some of which are very old. The concentration of these fish and mussels, however, makes them susceptible to exploitation. Recently, the high price of mussel shells in Japan has resulted in intensive mussel harvest and subsequent closure of the mussel season in Wisconsin inland waters.

Hydroelectric facilities that conduct peaking operations (varying flows to produce electricity for peak demand periods) have an effect on downstream habitats. The availability of stream habitat is largely a function of stream discharge (Trotzky and Gregory 1974, Milhous et al. 1981, Bovee 1982, Bain et al. 1988, Leonard and Orth 1988). Changes in discharge translate into changes in substrate, velocity, and depth conditions. These flow-dependent physical habitat features play an important role in governing the distribution and abundance of mussels (Salmon and Green 1983, Neves and Widlak 1987, Way et al. 1990, McMahon 1991, Strayer and Ralley 1993); consequently, hydroelectric peaking operations can influence the availability of mussel habitat by creating wide fluctuations in discharge. Erosion and sand and silt deposition have been implicated in decimation of mussel beds on the Mississippi River (Stansbery 1970). Recent surveys by David Heath (Wis. Dep. Nat. Resour., pers. comm.) indicate the only known population of winged mapleleaf mussel exists in the St. Croix River below the St. Croix Falls hydroelectric dam, where it is subjected to periodic exposure and desiccation due to water level manipulation.

Dams constructed to alter water levels on natural lakes can change the aquatic plant community. Large scale changes in aquatic plant communities, riparian and littoral zone habitat, and water quality have occurred at least in part because of these artificial water level manipulations. Changes in water levels following dam construction have destroyed wild rice beds on some waters (Vennum 1988). The Army Corps of Engineers has attempted to maintain a stable level in the Great Lakes in accord with an agreement with Canada; however, the wetlands, spits, and sand beaches of the Great Lakes are shaped by natural fluctuations in water level. The coastal marshes concentrate much of the biodiversity and productivity in the Great Lakes and short- and long-term lake level fluctuation cycles are critical for sustaining the plant communities (Keddy and Reznicek 1986, The Nature Conservancy 1994). When the operating levels of the Great Lakes were set, it is unlikely that consideration was given to the environmental features that would be affected. The level of Lake Winnebago, the state's largest inland lake, is also controlled by dams.

The construction of dams and the associated control of flood waters may affect the reproductivity of amphibians within the floodplain ecosystems of dammed rivers. In free flowing rivers, spring snow melts and rainstorms can add considerably to flow levels resulting in frequent flooding of lowland areas adjacent to the river corridor, providing added capacity for amphibian reproduction in the form of ephemeral ponds. Most of Wisconsin's amphibians require ephemeral, fishless ponds for reproduction (Vogt 1981). The hydroperiod of ephemeral waters has a direct influence on both the diversity and abundance of metamorphosing juvenile amphibians (Pechmann et al. 1991). In drought years especially, the input to ephemeral ponds from early spring snow melt and subsequent flooding may be essential for amphibian recruitment. Dams can and often do eliminate or minimize the opportunity for flood water to benefit amphibians. The ecological effect of reduced amphibian reproduction may be significant since amphibians generally represent high levels of biomass in deciduous forests (Burton and Likens 1975), a habitat often associated with floodplains. The creation of dams has also converted many seasonal wetlands to more permanent water within the reservoirs. This is especially evident on the Mississippi River. Although these flooded wetlands are more productive fishery waters, amphibian populations are reduced. The magnitude of losses of amphibian populations caused by flooding wetlands is unknown.

Extensive dam construction in Wisconsin has reduced the available habitat for riverine reptile populations, but the total impacts are unknown. Painted and snapping turtles, which normally occupy slow flowing or standing water environments, may displace riverine species like wood or map turtles in reservoirs. Impacts to amphibians by damming can also have direct impact on reptile species dependent on amphibians for food. For example, the diets of garter snakes and northern water snake consist primarily of frogs (Vogt 1981).

Aquatic insect communities in the presence of dams are qualitatively different and usually less stable than those in unregulated stream sections. The presence of an impoundment changes the habitat and quantity and quality of food released in downstream areas. Hydroelectric peaking operations result in large and rapid fluctuations in flows below dams (Cushman 1985) which can reduce species diversity, density, and biomass of aquatic insects in tailwaters, with certain taxa affected selectively (Fisher and LaVoy 1972, Trotzky and Gregory 1974, Williams and Winget 1979). Specific problems include increased drift rates, which are known to accompany extreme changes in flow (Radford and Hartland-Rowe 1971, Beckett and Miller 1982), and stranding of stream insects in "intertidal zones" as waters recede (Kroger 1973, Ward 1976, Extence 1981). Additionally, more time is required for aquatic insects to colonize habitats in rapidly varying flows than in unregulated flows (Gersich and Brusven 1981). Lentic insects have replaced lotic insects in impoundments resulting in net losses of lotic forms (Neel 1963, Hilsenhoff 1971, Ward 1976). Changes in energy processing in impoundments has usually led to substantial densities of collectors and collector-gatherers in tailwaters but low densities of shredders and predatory insects (Spence and Hynes 1971, Simmons and Voshell 1978).

Few new dams are being built at this time, but renovation and expansion of existing dams is common. The late 1980s expansion of the dam at Jim Falls in Chippewa County created the state's largest hydroelectric facility. Recent interest in renewable energy sources has lead to an increased number of hydroelectric development applications with the Federal Energy Regulatory Commission (FERC). Hydroelectric power is a "clean" energy source because it produces no air emissions or solid wastes. However, we do not have a complete understanding of the impact of dam construction on biological diversity in the affected river, although there is substantial evidence that modifications of the natural flooding and sediment transport cycles in river systems can dramatically simplify these systems. The Department may need to prepare to deal with the potential influx of hydropower development license requests.

Under current FERC regulations, hydroelectric facility owners/managers are required to give equal consideration to the resource as is given to power generation. This is a boost for environmental protection of riverine ecosystems, especially compared with past regulatory requirements for hydro facilities. The Department is obtaining valuable information about endangered and threatened species and working with hydro owners/managers to work out agreements to better protect the resources affected by their operations. Wherever possible these hydroelectric facilities are encouraged to go to a run-of-the-river flow regime in an attempt to reverse effects of past peaking operations. At a minimum, studies should be undertaken to determine the minimum levels of flow necessary to protect the flora and fauna of these rivers while still allowing hydro facilities to utilize this public resource. Some successes have been achieved, both through the regulatory process and by working cooperatively with the hydro owners/managers. The results are expected to benefit a variety of species, including mussels, other aquatic invertebrates, amphibians, and fish. Dam relicensing and regulation activities rarely consider abandonment as meaningful options, and funds to remove dams are limited.

Dam operation on the Mississippi River and associated commercial barge navigation continues to have impacts on that riverine ecosystem. Potential impacts include conversion of riverine habitat to lacustrine, modification of normal water levels, sediment resuspension, dredging and channelization, and increased recreational use (Holland and Huston 1984, Smart et al. 1985, Eckblad 1986, Holland 1987, Fremling et al. 1989). In recognition of some of these problems, the U.S. Congress established an environmental management program with the objective of restoring and monitoring habitat in the upper Mississippi River (Lubinski and Gutreuter 1993).

Dam construction has had many well-documented negative impacts on Wisconsin aquatic ecosystems, but it has also created additional reservoir habitat statewide. Balancing the widespread losses of riverine ecosystems with gains in lake habitat—of which Wisconsin already had a natural abundance—becomes a controversial proposition. Wildlife management activities that impound streams for waterfowl management often increase habitat for a variety of species, and have often been built on degraded or channelized wetlands.

However, such dams can still affect rivers and streams like any other dam. They may increase nutrient loading to the impoundment; disrupt movement of fish; change the character of existing wetlands from shrub, sedge, or wooded to predominately open water; and disrupt water and sediment movement. On a few lakes, the presence of large numbers of waterfowl leads to increased eutrophication through the deposition of their fecal material. Some flora, such as Fassett's locoweed, are intimately associated with specific lakes and their unique water level characteristics. Modifications of these fluctuations, changes in nutrient levels, or pesticide inputs from groundwater could threaten the existence of these plants.

> Many Wisconsin waters suffered severe simplification from the effects of industrial and municipal point-source pollution from the 1800s through the 1960s. Federal and state Clean Water legislation has led to dramatic improvements in water quality and the restoration of these aquatic communities. However, the accumulation of pollutants in sediments will remain a source of contamination to the biota for an extended period.

Point-Source Pollution

Many Wisconsin waters suffered severe simplification from the effects of industrial and municipal point-source pollution from the 1800s through the 1960s. Discharge of nutrient-rich sewage effluent reduces dissolved oxygen causing direct mortalities of fish and other aquatic organisms (e.g., Coble 1982). Discharge of toxic chemicals can also cause direct mortalities and lead to build-up of toxic materials in the aquatic system. Benthic invertebrate communities are simplified through loss of species sensitive to water quality and increased dominance of pollution-tolerant generalist species (Cuffney et al. 1984, Chadwick et al. 1986, Camargo 1992). Heavy metals and organic chemical pollutants can bioaccumulate in fish posing a threat to wildlife and human health (Kleinert et al. 1974).

Becker (1983) presents a discussion of this problem in Wisconsin which is otherwise not well documented. Paper and pulp mills concentrated along the Wisconsin and lower Fox Rivers were the major source of pollution discharging both nutrient-rich effluents and toxics such as mercury and polychlorinated biphenyls (PCBs). Untreated or poorly treated municipal sewage was a second major source of pollution in many river systems. Discharges of toxic heavy metals occurred in areas of heavy industrial development such as Milwaukee, Racine, and Kenosha counties, and in central Wisconsin (Konrad and Kleinert 1974). Impacts on Wisconsin's aquatic systems from point-source pollution have been severe in some areas. Aquatic life including fish and fish-eating birds suffered heavy mortality and reproductive impairment in the Wisconsin and lower Fox Rivers and in localized areas with heavy discharges (Becker 1983, Hauber 1989, Giesy et al. 1994).

Federal and state Clean Water legislation has led to dramatic improvements in water quality in these areas and major steps toward restoration of these aquatic communities. However, the accumulation of pollutants in sediments will remain a source of contamination to the biota for an extended period. Aquatic communities of the Great Lakes are particularly susceptible to substantial bioaccumulation of contaminants due to their long water-residence times. The approximate flush time in Lake Superior is 182 years; in Lake Michigan it is 106 years (Arimoto 1989).

Agriculture

Agriculture can have a dramatic impact on aquatic ecosystems. Aquatic systems are simplified by direct habitat destruction, erosion and sedimentation, hyper-eutrophication, and water quality degradation (e.g., Armour et al. 1991). Agricultural practices of particular concern are livestock grazing in riparian areas, plowing and tilling of erodible soils (particularly in areas of steep terrain such as the DRFT), concentrated nutrient runoff from barnyards and feed lots, pesticide and nutrient runoff from fields, loss of upland vegetation when forests and prairies are brought under cultivation, dredging and filling of wetlands, and channelization of streams. Almost all the agricultural chemicals in use are water soluble, resulting in high mobility by water transport and thus a significant water pollution problem with the potential for chronic effects on aquatic organisms (Sagar 1991).

> Agriculture can have a dramatic impact on aquatic ecosystems. Practices of particular concern are livestock grazing in riparian areas, plowing and tilling of erodible soils, concentrated nutrient runoff from barnyards and feed lots, pesticide and nutrient runoff from fields, loss of upland vegetation when forests and prairies are brought under cultivation, dredging and filling of wetlands, and channelization of streams.

Agricultural impacts on aquatic organisms in Wisconsin and other Midwestern states are also well documented. Karr et al. (1985) estimated that 44% and 67% of fish species have disappeared or become less abundant in major Ohio and Illinois river systems and cited agricultural pollution as having had the greatest impact. Erosion and sedimentation have degraded many stream channels, resulting in severe impacts to these and downstream aquatic communities. Sedimentation profoundly changes stream insect populations (Rosenberg and Wiens 1978, Newbold et al. 1980, Lemly 1982, Culp et al. 1986). Paleolimnological evidence from Lake Mendota suggests there was a dramatic increase in sedimentation and eutrophication after 1800, when agriculture began in the basin (Kitchell and Sanford 1992). Biological communities also became more unstable, suggesting increased perturbation of the aquatic community.

One of the rarest fish in the state, the bluntnose darter, may have been affected by increased siltation due to plowing of the prairies (Pflieger 1971). This species prefers quiet oxbows, ponds, and sloughs with mud, clay, and mixed sand and mud bottoms. The population of mud darter, another rare fish in Wisconsin, declined in Illinois, due to decreased river size and reduced flows (Smith 1968). Decreased river size and flows in Wisconsin could occur due to groundwater pumping, pumping for agricultural irrigation, or droughts. Greene (1935) recorded the least darter in southeastern Wisconsin but recent surveys (Fago 1992) have not found the species there. According to Becker (1983) this loss may be due to increased turbidity and habitat destruction caused by agricultural, domestic, and industrial pollutants.

Specialist fish have been the most severely impacted. For example, Becker (1983) notes the gravel chub is limited to the lower Rock River drainage of Wisconsin and states, “the habitat requirements of the gravel chub are so strict that populations are isolated and confined to special riffle areas with special bottom types.” This specialization has made it vulnerable to turbidity and siltation, which increased as a result of agricultural activities. The creek chubsucker has probably been extirpated from the southeastern part of the state, where it was at the northern end of its range in the Des Plaines River (Becker 1983). Becker (1983) believes erosion and habitat destruction in the watershed eliminated the remnant population of the creek chubsucker by the middle part of the twentieth century.

The Ozark minnow is noted by Becker (1983) to be absent from a number of locations where it was previously reported, apparently because it is intolerant of excessive turbidity and siltation. Most of the streams where the Ozark minnow was located are characterized by heavy agricultural use. Becker (1983) also reports the pugnose shiner, a state-threatened species, does not tolerate turbid conditions.

The state-endangered queen snake has also been impacted by erosion and sedimentation resulting from agriculture in southeastern Wisconsin. This species has a very specialized diet consisting almost exclusively of crayfish (Vogt 1981) and requires a micro-habitat consisting of flat rocks on the stream bed under which it forages and seeks cover (Wood 1949). Many of the streams once utilized by queen snakes have experienced heavy sedimentation resulting in a loss of exposed rocky stream bed and an associated reduction or loss of crayfish populations (Gary Casper, Milwaukee Public Museum, pers. comm.).

Agriculture also affects amphibian populations in more ways than just by eliminating or altering their critical breeding and foraging habitats. Frogs and salamanders have very thin, permeable skin and are vulnerable to chemical alterations of their terrestrial and aquatic environments. The eggs and larvae are especially susceptible. Amphibians are considered to be excellent indicators of environmental health. Extremely high mortality and developmental abnormalities for some species are the result of toxicity caused by agricultural chemicals in aquatic systems (Hazelwood 1970, Birge et al. 1980). The Blanchard's cricket frog, Wisconsin's most endangered amphibian, has seen a dramatic decline throughout its historic range (Minton 1972, Christoffel and Hay, Wis. Dep. Nat. Resour., unpubl. data). While no specific cause has been implicated, it is suspected that agricultural chemicals (e.g., atrazine) are, in part, responsible for this decline. Hylid frogs in general, such as the cricket frog, may be more susceptible to pesticides than other frog species (Sanders 1970, Birge et al. 1979). These agricultural impacts may also be magnified through bioaccumulation in amphibian prey sources (Hazelwood 1970, Sanders 1970, Birge et al. 1980, Hall and Kolbe 1980, Linder et al. 1990).

Non-Agricultural Nonpoint-Source Pollution

Not all nonpoint-source pollution comes from agriculture; it also results from urban stormwater runoff, use of fertilizers and chemicals in urban areas, construction site erosion, poorly designed or leaking septic systems, and poor land management practices in non-agricultural developments.

The U.S. Environmental Protection Agency (EPA) estimates that 50% of water pollution in the U.S. is from nonpoint sources (Barton 1978). A 1985 survey indicated that 36% of all Wisconsin's streams and rivers are affected to some degree by nonpoint-source pollution (Bergquist 1986*a*). Not all nonpoint-source pollution comes from agriculture; it also results from urban stormwater runoff, use of fertilizers and chemicals in urban areas, construction site erosion, poorly designed or leaking septic systems, and poor land management practices in non-agricultural developments. Surface nonpoint pollution can include nutrient runoff, erosion and sedimentation, and toxic substances. Loss of terrestrial

Nutrients from nonpoint pollution enter lakes and are recycled during spring and fall turn-over. Excessive plant growth and algae are often the result. *Photo from DNR files.*

vegetation in urban areas increases the amount and variability of runoff events contributing to flooding and erosion in downstream areas.

The addition of nutrients from nonpoint sources increases the nutrient loading of the lakes and artificially accelerates the eutrophication process. Once a lake is overloaded with nutrients, they are hard to remove, since the nutrients are continually recycled during spring and fall overturn. Increased nutrients cause increased algae or macrophyte growth. Excessive increases in plant growth are often dominated by a few species reducing aquatic plant species diversity. The proliferation of macrophytes into the entire euphotic area of the littoral zone leads to loss of small openings for fish spawning and creates an extreme amount of escape cover for young-of-the-year fish, which can become overpopulated and stunted. The resulting competition for limited food resources can adversely affect fish species and benthic organisms that may be either a food source or a competitor for food. Decay of the increased plant biomass when it dies can result in decreased dissolved oxygen levels and kills of fish and other aquatic organisms.

Changes in Wisconsin's aquatic systems caused by non-agricultural nonpoint-source pollution are less well documented than in agricultural areas, probably because they have been isolated in highly urban areas and masked by point-source and agricultural pollution problems. Since intense urbanization is a relatively recent phenomenon in most of Wisconsin, it is probable that urban nonpoint-source pollution has only recently been impacting aquatic ecosystems on a statewide scale. Except in a few isolated watersheds, rural and urban nonpoint-source problems have not been controlled. The state's major nonpoint-source abatement activity is the Priority Watershed program (Bergquist 1986*b*). The effectiveness of this program in achieving results has been questioned, and evaluation efforts have only recently been initiated (Simonson and Lyons 1992). New laws requiring storm water retention basins in new developments will help but do not address problems from existing development.

Contamination of groundwater and surface waters from abandoned landfills and leaking underground storage tanks continues. Inventory of these sites is incomplete, and their contents are often not known, but many may contain hazardous and toxic materials. The amount of contamination depends on the rate at which the site fails, the content of the site, its proximity to the aquatic resource, and the soils and geology of the area. However, since the groundwater gradients are generally in the direction of surface waters, it will only be a matter of time before the contaminated groundwater reaches a surface water.

Poorly designed and leaking septic systems can lead to water quality problems in unsewered residential areas. Lakefront development is of particular concern because of its proximity to surface waters and higher than normal density of septic systems. Lakefront developments are often in rural areas where connection to sewer systems is very costly. Elimination of nutrient inputs to lakes often does not improve water quality because previously added nutrients are concentrated in lake sediments and continuously resuspended and recycled.

Timber Harvest

The impacts of silvicultural activities in Wisconsin on water quality are not well studied. Timber harvest within watersheds and along riparian areas has been shown to affect water quality in other regions of the country through increased runoff, sedimentation, and temperature, and by reducing primary productivity and dissolved oxygen (e.g., Gray and Edington 1969, Hibbert 1969, Fredrickson 1970, Hornbeck et al. 1970, Hansmann and Phinney 1973, Beschta 1978, Pearce and Rowe 1979, Bernath et al. 1982, Hewlett and Fortson 1982, Lynch et al. 1984, Noel et al. 1986, Verry 1986, Hicks et al. 1991).

Large woody debris normally resulting from streamside bank erosion or blowdowns plays an important role in stream and river morphology, hydrology, and ecology. Bilby and Ward (1989) studied the relationship of woody debris to the size of streams in western Washington. Large pieces of woody debris influenced channel morphology there through bank erosion, channel scouring, deposition, sandbar formation, nutrient and organic material retention, and species composition. However, the larger the river, the larger the woody debris needed to overcome the capacity of the river to move the debris downstream. The mean diameter, length, and volume of woody debris increased as channel width increased. Murphy and Koski (1989) studied the rate of input and depletion of large woody debris in Alaskan streams. They found the rate of input and depletion was inversely proportional to the diameter of the debris. The model used predicted that 90 years after a clear cut, large woody debris would be reduced by 70%, and it would take 250 years to return to prelogging levels. They recommended a 30-m wide unlogged buffer strip next to streams to maintain large woody debris for input to streams. Benke et al. (1985) showed that although woody debris accounted for only 4% of habitat surfaces in a low gradient Georgia coastal stream, they supported 60% of the invertebrate biomass and 16% of the production for a river reach. Losses of habitat elements such as large woody debris can have effects for 80 to 160 years (Sedell and Frogatt 1984, Sedell and Swanson 1984, Minckley and Rinne 1985).

Although many of these studies are not specific to Wisconsin, the relationship between water quality and logging practices is important. Given the historical intensity of timber harvest in northern Wisconsin, it is likely that some forestry practices have had similar water quality and habitat reduction impacts in Wisconsin's aquatic systems. For example, Watermolen (1993*a*) lists some specific streams in the upper Green Bay basin that have been impacted by recent forestry practices.

Large woody debris such as this fallen tree plays an important role in stream and river ecology. *Photo by Betty Les.*

While most public lands have aesthetic management zones to maintain the visual appeal of an undisturbed shoreline, harvesting practices on the backlands can still lead to erosion and disruption of overland water flow. Wisconsin has developed a new program, Wisconsin's Forestry Best Management Practices for Water Quality, which will help address these concerns.

Channelization and Clearing of Streams

Streams have been straightened or channelized in the mistaken belief that hydraulic efficiency was better for the conveyance of flood waters brought on by runoff from pastures and intensively farmed cropland and denuded forest lands. Removal of natural obstructions to navigation have also been commonplace, particularly during the period when rivers were extensively used to transport logs. Channelization is known to reduce species richness and diversity in fish, aquatic invertebrates, and mussels, and to impact other organisms such as furbearers that depend on aquatic systems (Schneberger and Funk 1971, Yokley and Gooch 1976, Yokley 1977, Arner et al. 1979, Schlosser 1982, Kanehl and Lyons 1992). Further, it can often lead to the instream disposal of dredge spoils which is detrimental to aquatic life. Instream disposal directly affects fish reproduction, benthos and water quality (Morton 1977). The channelization and clearing of streams has eliminated entire reaches of valuable aquatic communities for warm water, cool water, and cold water species. Reduced amounts of large woody debris in streams can alter aquatic insect community structure, especially in rivers with a shifting sand bed (Dudley and Anderson 1982; Benke et al. 1984).

Channelization often results in the reduction of natural edge along aquatic corridors and also results in the disturbance of shoreline vegetation, opening the door for invading exotic plant species. Many of Wisconsin's herptile species rely on these riparian areas for a great deal of the active season for shelter and foraging (Vogt 1981). Channelization does not likely threaten most herptile populations but it is certain their numbers are reduced by it.

The establishment of exotic species or hybrids in an aquatic ecosystem may initially appear to increase species richness and diversity. However in the long term, invasions of exotics may result in the loss of native species and the disruption of habitats, predator-prey relationships, and energy flow processes.

The extreme impacts of channelization and dredging in Wisconsin's waters have been well documented. While these activities have been curtailed, permits are still sometimes issued. Smaller development or maintenance projects are still permitted by the state when the local regulator does not believe the environmental impacts outweigh the perceived benefits. Large navigation projects such as the Mississippi River are under federal control.

Invasion of Exotic Species

The establishment of exotic species or hybrids in an aquatic ecosystem may initially appear to increase species richness and diversity. However in the long term, invasions of exotics may result in the loss of native species and the disruption of habitats, predator-prey relationships, and energy flow processes. Exotic species often invade without the normal predators or parasites that control their numbers in their native ecosystems, and existing ecosystems may be unable to accommodate the new species without an overall simplification of the community. Introduced exotics are often disturbance-tolerant, hardy generalists having successfully survived human introductory mechanisms such as overseas shipping or passage through pre- and post-export chemical treatments. These hardy species are well adapted to exploit already stressed and over-simplified biotic communities. In some cases, exotics initially explode in numbers but eventually stabilize at a lower level of abundance. It is difficult to predict the impact that a new exotic species will have on an existing aquatic ecosystem.

Non-native species have frequently invaded or have been introduced to Wisconsin's aquatic communities. Several key exotics unintentionally gained access to the Great Lakes via the St. Lawrence Seaway, either transported in ballast water, attached to vessels (Moyle 1991), or by direct migration. Invader species include the Asiatic clam, the sea lamprey, river ruffe, white perch, *Bythotrephes* (a predatory cladoceran), and the zebra mussel. Other species have been intentionally stocked, including the common carp, which was brought in with the best of intentions in the late 1800s. The introduction of this species is the most infamous example of a management action that was thought to be beneficial at the time—but turned out to have devastating consequences (Courtney and Moyle 1992) which managers are still struggling to cope with today. Introduction of desirable species such as brown and rainbow trout have had unknown impacts on native brook trout. The grass carp, a more recent introduction, is now reproducing in the lower Mississippi River.

Exotics can also be introduced through releases of species used for bait. There have historically been few controls or monitoring of the harvest, transfer, or sale of fish or aquatic invertebrates used for bait (Threinen 1982). It is believed the rusty crayfish was accidently introduced from Illinois by bait anglers. The impacts of rusty crayfish on the communities in certain waters have been great (e.g., Olsen et al. 1991).

There are several well documented invasions of exotics in Wisconsin. Eurasian water milfoil was first discovered in Wisconsin in 1967 and has now spread to at least 75 lakes in 39 counties (Bode et al. 1993). The explosive growth of this plant can substantially alter native aquatic plant communities, interfering with recreational use, impacting fish communities, and choking water intakes.

The river ruffe is already the second most abundant species in the St. Louis River estuary, and biologists fear that it is a predator of whitefish eggs and that it can successfully compete against yellow perch (Moyle 1991). The white perch now found in Green Bay also has the potential to overtake the native yellow perch; however, several studies have not found such impacts in Oneida Lake, New York, or Lake Erie (Forney 1974, Schaeffer and Margraf 1987). Fuller (1974) considers the Asiatic clam to be a form of pollution itself. This species is a threat due to its free-swimming larva and its ability to exploit any available substrate though there is no evidence to indicate that the Asiatic clam can successfully compete against other clams and mussels in Wisconsin as it does in some southern states.

The zebra mussel has become established in Lake Michigan and the Mississippi River, and its numbers have significantly increased to date. This invader poses a significant threat to native mussels. Native mussel populations have already declined in some areas of the Great Lakes Basin due to the impacts of zebra mussels (Hebert et al. 1991; Mackie 1991). The potential impacts of zebra mussels on native bivalve populations have important implications for the upper Mississippi River, which has one of the most rich and diverse mussel populations in the world (Cope, U. S. Fish and Wildl. Serv., unpubl. data). In addition, zebra mussels have been identified as responsible for concentrating organochlorine pollutants and maintaining them in the food chain (Stone 1994).

The impacts of exotics on Wisconsin's aquatic ecosystems are difficult to assess. Intentional introductions of brown and rainbow trout, Pacific salmon, striped bass, and grass carp are often cited as examples of successful introductions of non-native species, but the long-term implications of these introductions are poorly understood. No exotic that has become established has ever been eradicated, so the risks associated with introducing exotics are extremely high. It is unlikely that any species have been extirpated from Wisconsin because of exotics, but it is probable that native species such as brook trout have been significantly reduced in abundance and distribution by competition from exotics (e.g., Waters 1983, Larson and Moore 1985). The invasions of carp, river ruffe, sea lamprey, alewife, zebra mussels, white perch, rusty crayfish, purple loosestrife, and Eurasian water milfoil have already had negative impacts on native ecosystems. Control of these invasions is already beyond the capability of any management agency. Management agencies across the country, however, continue to propose introduction of new exotics. Most states around Wisconsin have already allowed introduction of the grass carp for control of aquatic macrophytes with supposed safeguards against their becoming naturalized. Despite these safeguards, grass carp have successfully reproduced in the lower Mississippi River (Allen and Wattendorf 1987). Well-intentioned introductions of largemouth and smallmouth bass in Texas have led to genetic introgression with the endemic Guadalupe bass, which is now well on the way to extirpation in some river systems (Morizot et al. 1991).

Introductions of supposedly infertile sauger-walleye hybrids and stocking of sauger into native walleye waters have led to genetic introgression between the two species (Billington et al. 1988). Sauger-walleye hybrids have been stocked in some Wisconsin waters. North Dakota has already stocked zander, a close European relative of the walleye, into supposedly landlocked waters in the state (Terry Steinwand, pers. comm.). Its escape and establishment in native walleye waters would undoubtedly have a devastating impact on native walleye and sauger populations.

New regulations to control the introduction of exotics through Great Lakes ballast water exchanges have been proposed by the U.S. Coast Guard. Wisconsin has adopted new laws to eliminate the importation of exotic fish species. The Department has not allowed the use of grass carp and has taken steps to actively eliminate populations that were discovered. The Department has not proposed stocking other exotic species in recent years. In addition, the Department has undertaken public education programs designed to minimize spread of exotic organisms such as Eurasian water milfoil and Zebra mussel, and participated in monitoring programs for species such as purple loosestrife and Zebra mussel. Whether these actions will be sufficient to prevent the continued introduction of exotic species into Wisconsin waters is unknown, but recent history suggests far more rigorous efforts may be needed.

Left undeveloped, riparian areas provide food and habitat for species such as this great blue heron. *Photo by Bob Queen.*

Riparian Development

Riparian habitat along Wisconsin's lakes and rivers has been extensively developed since the mid-1800s. Shoreline development in populated southern areas occurred early in this period where people lived and worked. Development on northern lakes and rivers came later and was initially limited by the remoteness of the area and its sparse population. However, as wages and leisure time increased, as transportation improved, and as the state's population grew, more people were interested in second homes. Cottages, resorts, shacks, trailers, and all manner of dwellings were built. Before zoning laws existed, some structures were built only a few feet from the shoreline; trees and logs were cleared from both water and shore, and privies or septic systems were put in. The level land and sandy shorelines suitable for beaches on well-known lakes disappeared first, followed by development on less desirable land that was steeper, rockier, or marshier. Rates of development have continued to escalate in recent years. The number of lake front homes in Forest County, for example, has increased 700% during the last ten years. In the Brule area, development has increased 19% for 200-450 acre lakes and 78% for 100-124 acre lakes over the past 10-30 years (Korth 1993). Only a few isolated lakes escaped extensive development, including some large flowages such as the Turtle-Flambeau, Chippewa, Gile, Rainbow, and Willow, as well as small lakes or lakes on land owned by paper companies or public agencies such as the U.S. Forest Service, counties, and the Department.

With riparian development came extensive loss and simplification of aquatic habitats. Owners of lake property commonly modified the shoreline or littoral area adjacent to their property by using sand blankets, shoreline protection such as riprap and retaining walls, docks and piers, boat houses, dredging for access, aquatic plant nuisance control, and filling. Disruption of natural shoreline changed gradations in water depth in lakes, thereby eliminating natural formation of plant communities (Keddy 1983), and similar development along streams causes changes in the structure of the macroinvertebrate communities (Cummins et al. 1984, Sweeney 1993). Aquatic plant communities were frequently directly altered through mechanical removals or chemical treatments. Many alterations were done by specific riparian property owners, but some municipalities have operated large-scale aquatic plant control activities.

Isolated cases of shoreline modification may have little potential for affecting the aquatic community, but the cumulative effects of numerous alterations can have significant and long-lasting impacts due to habitat loss and simplification (Panek 1979). Some Wisconsin lakes, such as Shawano Lake, have very little natural shoreline left. Lyons (1989*b*) reported a significant simplification of the littoral zone fish community of Lake Mendota since 1900 and attributed the changes in part to increased shoreline development. Bryan and Scarnecchia (1992) found significantly fewer fish species and reduced abundance in developed shoreline areas in an Iowa lake. Miller et al. (1989) conclude that habitat alteration was a factor in 73% of fish extinctions in North America during the past 100 years. Kapuscinski and Jacobson (1987) explain that habitat alteration can impact genetic diversity by reducing the effective population sizes and changing selection pressures on previously well-adapted species.

Isolated cases of shoreline modification may have little potential for affecting the aquatic community, but the effects of numerous alterations can have significant and long-lasting impacts due to cumulative habitat loss and simplification.

Lakes were also affected by the draining and filling of wetlands, which supported waterfowl, reptiles, and amphibians, northern pike, and muskellunge spawning areas. Loons, ospreys, eagles, otter, muskrat, and mink were all affected by habitat degradation and harassment resulting from increased use. Habitat loss and urbanization have been implicated in reducing populations of several dragonfly species that are threatened or endangered in Wisconsin (Nilles 1993).

The cumulative effects of numerous shoreline alterations may be detrimental to local amphibian and reptile populations (Watermolen 1993*b*). Many amphibian species dependent on the shoreline/water interface (e.g. green, mink, and bull frogs) are displaced when seawall construction replaces the natural shoreline. The greater the loss of natural shoreline the greater the impact to the local frog population. Aquatic turtles, which need to leave the water to lay their eggs on land, are affected by shoreline barriers. Several species of Wisconsin turtles show strong signs of nest site fidelity. When turtles are prevented immediate access to these sites because of shoreline development, they are forced to expend additional time and energy searching for a new site or travelling indirectly to their traditional site. This exposes them to potentially higher mortality since most aquatic species have little natural defense on land. What effect this has on populations is unclear.

Despite the well-documented negative impacts of riparian development, it continues on Wisconsin's waters at a rapid pace. There are few legal restrictions to development, a situation difficult to change because of the high demand for and value of lakeshore property. The Department controls permitting of erosion control structures and lake bed modifications, but such decisions are increasingly being challenged in legal forums. Increasing evidence also suggests that riparian activities beyond the ordinary high water mark, which are not controlled by the Department, have impacts on the systems. Research has been done in Canada to predict sustainable levels of lakeshore development (Dillon and Rigler 1975), but these methods have not been applied in Wisconsin. Filling of riparian wetland areas has been dramatically curtailed, but is still a concern.

Riparian areas are also increasingly impacted by beaver activities. Wisconsin beaver populations have been increasing in recent years due to elimination of natural predators, reduced trapping pressure, and habitat management practices that increase aspen and willow. Beaver activity can have a significant impact on riparian areas, including damming of streams, cutting of trees, and flooding of low-lying areas. While moderate levels of beaver activity are probably necessary to maintain native habitat patterns along streams, excessive levels are thought to disrupt fish movement, alter sedimentation patterns, and increase water temperatures, adversely affecting cold-water communities.

Fish have been artificially propagated and widely stocked in Wisconsin for more than a century. Fish have also been routinely moved from one water body to another, and new species have been widely introduced . . . . While these efforts reflected the best understanding of fish management at the time, there is growing evidence that stocking or transfers of fish can have long-term negative impacts on growth, survival, reproduction and even health of both the existing fish population and the newcomers.

Fish Stocking and Poor Understanding of Genetic Diversity

Fish have been artificially propagated and widely stocked in Wisconsin for more than a century. Fish have also been routinely moved from one water body to another, and new species have been widely introduced. The magnitude of the fish-stocking and transfer program in Wisconsin since 1874 is staggering. Virtually every major lake and river has been stocked at one time or another by the Department or private individuals. Becker (1983) presents an excellent summary of the evolution of the Wisconsin stocking program. By 1900, Wisconsin had attempted introductions of Atlantic salmon, chinook salmon, grayling, rainbow and brown trout, carp, and goldfish. Walleye propagation began in 1883 with production of eight million fry. Muskellunge propagation began in 1897 with production of one million fry. Hatching and stocking of lake trout, brook trout, and whitefish was done as early as 1876. In 1937, Wisconsin stocked over a billion fish of various species.

Currently the Department annually stocks on average about 615,000 brook trout, 2.2 million brown trout, 940,000 rainbow trout, 280,000 lake trout, 260,000 splake (brook trout x lake trout hybrids), 2.3 million chinook salmon, 659,000 coho salmon, 513,000 largemouth bass, 63,000 smallmouth bass, 169,000 muskellunge, 28,000 hybrid muskellunge (northern pike x muskellunge hybrids), 60,000 northern pike, 2,300 lake sturgeon, 3.5 million walleyes as fingerlings, yearlings, or catchable-sized fish. Another 65,000 largemouth bass, 850,000 muskellunge, 344,000 hybrid muskellunge, 15 million northern pike, 61,000 lake sturgeon, and 51 million walleyes are stocked as newly hatched fry. Other species stocked periodically include channel catfish, sauger and several species of panfish (Dave Ives, Wis. Dep. Nat. Resour., pers. comm.).

Trapping and transfers of adult fish was common from 1874 until the 1930s (Becker 1983). Fish were often “rescued” from waters expected to experience winterkill or from flooded backwater areas of the Mississippi River as they were stranded by receding flood waters in the spring. Rescued fish were transferred to waters across the state. In 1936, nearly ten million fish, including catfish, sunfish, crappies, bass, and buffalo, were trapped and transferred among various state waters. This activity became less common after 1940 but is still used in some winterkill and panfish management situations.

While these efforts reflected the best understanding of fish management at the time, there is growing evidence that stocking or transfers of fish can have long-term negative impacts on growth, survival, reproduction and even health of both the existing fish population and the newcomers. Largemouth bass moved between Texas, Florida, Wisconsin, and Illinois invariably showed significantly lower growth and survival in non-native waters (Philipp and Whitt 1991, Philipp 1991). Cutthroat trout stocking in Yellowstone was shown to have disrupted and reduced natural reproduction (Gresswell and Varley 1988). A comprehensive literature review examining releases of cultured salmonids into native populations concluded that, when effects were seen, they were always detrimental to the native stocks (Hindar et al. 1991). Hybridization, often resulting from stocking of different genetic strains, was a factor in 38% of fish extinctions in North America during the last 100 years (Miller et al. 1989). Hatchery fish can introduce poorly adapted genomes into the population and through introgression disrupt the genome of the existing naturally reproduced fish (Magnuson 1976). Introductions of different genomes caused by releases of bait fish can have the same effect.

Genetically different stocks may exist for many important fish species found in Wisconsin (Kapuscinski and Lannan 1986). Analysis of DNA pattern variations in walleye suggest that current stocks evolved from three distinct glacial refugia, and different walleye genetic types show clear regional distribution patterns (Billington and Hebert 1988). Genetic differences among walleye stocks have been documented within states (McInerny et al. 1991) and even within the same drainage (Todd 1990). Similar differences have been documented for largemouth bass (Philipp et al. 1983) and northern pike (Seeb et al. 1987). Only limited work has been done to analyze genetic variability among stocks in Wisconsin. The Department is currently funding a study of genetic differences among different spawning populations in the Lake Winnebago system. Preliminary results show limited allozyme variability (Treloar and Ehlinger 1991). The Department has also recently begun a major effort to quantify the genetic differences among watersheds for an additional eight warm-water species and native brook trout.

Fish stocking and transfers can also impact biodiversity by introducing new species into aquatic communities, with resulting changes in the relative abundances of the native species. Walleye, for example, is not native to small seepage lakes in Wisconsin (Becker 1983), but this species is now found in almost every such lake more than 200 acres in size. The impact of these introductions on the existing aquatic community is not well understood, but Colby et al. (1987) document fish community changes resulting from species introductions. In Wisconsin, for example, introduction of walleye and northern pike into Escanaba Lake resulted in long-term declines of smallmouth bass and panfish populations (Kempinger et al. 1975).

Release of bait fish and macroinvertebrates is another source of genetic mixing. Bait species are commonly harvested from naturally occurring populations and transferred to other waters for sale and potential release. Cultured bait species are also commonly sold but suffer the same potential genetic disruption as that caused by any other hatchery-reared species. The bait industry in Wisconsin is lightly regulated and little information is collected on the origin of fish that are sold.

Some species, such as walleye and lake trout, exhibit homing instincts during spawning. Walleye and lake trout also seem to exhibit spawning preferences and requirements unique to specific strains. Whether stocking of hatchery-reared fish has already affected this behavior is not known. In addition to the genetic effects of stocking, stocking large numbers of hatchery-reared fish among relatively few naturally reproduced fish can subject the natural population to increased fishing pressure because fishing gear is nonselective.

The actual impacts of fish stocking and transfer activities on Wisconsin's aquatic communities can never be fully known. However, given the magnitude of the number of fish and waters stocked, and the likelihood of genetic and population changes, it is probable that there have been significant changes in species abundances and distributions across the state. Becker (1983) suggests many anomalous distribution records may be due to these activities. Certainly the distribution of some major game species such as walleye, muskellunge, brown trout, and rainbow trout have been dramatically increased, but it is also likely that species such as largemouth or smallmouth bass and brook trout that existed prior to the introductions have suffered a corresponding decline in abundance and distribution. Changes in lower trophic level species are undocumented. Genetic impacts of stockings of species in waters where there were already naturally reproducing populations of those species are unknown, but it is likely that some populations have suffered declines in natural reproduction.

This evidence has led to numerous recommendations against mixing different genetic stocks. After an extensive survey of salmonid stocking effects, Hindar et al. (1991) recommend "strong restrictions on gene flow from cultured to wild populations and effective monitoring of such gene flow." In a reference work on fisheries genetics, Kapuscinski and Jacobson (1987) recommend managing to avoid stocking and, when necessary, stocking only locally adapted fish. Both Meffe (1987) and Kapuscinski and Philipp (1988) conclude that significant problems in conserving existing levels of genetic diversity exist and that additional cooperation and research will be needed to determine appropriate management strategies. The American Fisheries Society has developed a draft position statement entitled "Protecting Native Fish Stocks: The Elimination of Stock Transfers," which advocates a stock concept of management and restrictive stocking and stock transfer policies designed to protect native stocks (Philipp et al. 1991).

Population changes caused by long-term or size-selective harvest have often been shown; however, genetic changes are poorly documented (Policansky 1993). Highly selective gillnet fisheries were implicated in long-term changes in lake whitefish growth rate, condition factor, and mean age (Handford et al. 1977). High exploitation rates have changed growth and mean age in lake whitefish and walleye populations (Healey 1980, Reid and Momot 1985, Mosindy et al. 1987). Nuhfer and Alexander (1991) conclude that long-term angling exploitation may alter the genetic composition of wild brook trout strains.

Stocking and stock transfers remain important management practices in Wisconsin. Adult stock transfers of northern pike and various panfish species between waters and watersheds still occur. Current Wisconsin hatchery practices and stocking policies do not consider genetic conservation. For example, virtually all walleye and muskellunge hatchery production comes from the Spooner and Woodruff hatcheries—both located in far northern Wisconsin. All spawn is taken from local lakes, but the hatched fish are distributed throughout the state. Wisconsin also periodically exchanges hatchery products with other states and federal agencies.

Today, managers are increasingly less interested in stocking hatchery-reared fish and more interested in depending on the native stock for reproduction. Where stocking is needed, attempts are being made to use native brood stock. Programs are being initiated to study genetic diversity of fish within the state.

Harvest

Humans have been harvesting Wisconsin's aquatic resources for thousands of years, and this harvest has undoubtedly had impacts on biodiversity. There is considerable evidence that Native Americans used nets and spears and even built dams and weirs to harvest fish (Kuhm 1928) and turtles (Adler 1968). European settlers in the mid-1800s began commercially harvesting sturgeon, lake trout, suckers, yellow perch, and other common fish species.

Today, sport, commercial, and subsistence harvest of some aquatic organisms is substantial. Sport harvest activities are primarily directed at game fish species. During 1991, 1.47 million licensed anglers and more than 2-million anglers over-all fished 21.3 million days in Wisconsin. Over 0.5 million anglers were non-residents—second only to Florida in fishing by tourists. Anglers most frequently fish for panfish (bluegill, pumpkinseed, yellow perch, rock bass) followed by walleye and sauger, largemouth and smallmouth bass, northern pike and muskellunge, and crappie (U.S. Dep. Int. / U.S. Dep. Comm. 1993).

Subsistence harvest is practiced by many users to some extent when fish or other sport harvested animals are eaten. However, the only significant subsistence harvest of aquatic organisms currently allowed is a Native American walleye and muskellunge spear fishery. This fishery was reinitiated in 1986 and has been conservatively regulated and extensively monitored (Hansen 1989, Staggs et al. 1990, Hansen et al. 1991, U.S. Dep. Int. 1991).

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Commercial overfishing has been directly implicated in the decline of lake sturgeon, certain cisco species, Great Lakes brook trout, and lake trout in Wisconsin waters, although dam construction, habitat losses, and introduction of exotic species were undoubtedly also factors in the decline of these species (Becker 1983).

During this century, regulated sport angling has been the dominant harvest method for most Wisconsin fish. The effects of angling harvest on the most sought-after species are fairly well understood. Non-selective harvest typically increases total mortality and reduces abundance, but anglers generally select for larger fish. Such size-selective harvest can also reduce the relative abundance of older and larger fish, lower the average age of first reproduction, increase growth rates, increase the variability in recruitment (e.g., Spangler et al. 1977, Coble 1988, SPOF 1983), and may alter the genetic composition of a stock (Policansky 1993). Despite these changes, there is little evidence that sport angling alone can collapse sport fish populations. Unrestricted angling since 1946 in Escanaba Lake has not decreased the walleye population (Staggs et al. 1990). Angling has not been cited as a major factor in the extirpation of any fish in Wisconsin (Becker 1983) or extinction of any species in North America (Miller et al. 1989). Careful monitoring of this activity and its impacts, however, remains vital.

The indirect effects of angling harvest on other species in the aquatic community are less well understood. Nonharvested species are likely to be affected by changes in predation intensity or food availability when game or commercial species are harvested. Species interactions in north-temperate fish communities are complex and often affected by weather or other unpredictable factors (Colby et al. 1987). Under these conditions, demonstrating that changes in nonharvested species are caused by harvest of other species is difficult. Changes in relative abundance of fish

Fishing has been a popular sport in Wisconsin for many years. This photo from the 1930s shows a day's catch of muskellunge from Pine Lake in Vilas County. *Photographer unknown.*

targeted by sport angling or commercial harvest can have measurable impacts on the absolute and relative abundance of plankton species, which in turn can have impacts on nutrient cycling and water quality (e.g., Brooks and Dodson 1965, Shapiro et al. 1975, Kitchell 1992).

Commercial harvest of fish is regulated and information on the harvested stocks suggests that impacts are similar to those caused by sport angling. However, management programs for commercial fisheries on a national and international scale have more frequently than not failed to prevent collapsed stocks (e.g., Ludwig et al. 1993) and in past years commercial fishing here in Wisconsin has contributed to collapsed and extirpated Great Lakes fish (Becker 1983), so these fisheries must be managed conservatively (Peterman and Bradford 1987, Peterman 1990).

There has also been a significant commercial harvest of small fish, aquatic insects, and crayfish for bait sale. It is generally assumed that impacts on the harvested species have been minimal because these species typically have high fecundity or are widely distributed relative to the areas of harvest. However, these fisheries are lightly regulated, and very little information exists on the number of organisms harvested, much less the impact on the aquatic ecosystems.

There is growing evidence that harvest of herptiles is reducing the populations of some species in the United States (Dundee et al. 1992; James Harding, Mich. State Univ., pers. comm.). Changes in status of Wisconsin herptile species caused by harvest are not well known. The recent surge in the herptile pet trade has increased the exploitation of some Wisconsin species and even the state-threatened wood turtle is being smuggled as more states add this species to their protection list, creating a greater demand on protected stocks (James Harding, Mich. State Univ., pers. comm.). Wood turtles are popular turtles for the pet trade in the U.S. and abroad. Snapping turtles have been trapped and hooked for many years in Wisconsin. The pools of the Mississippi have been extensively trapped commercially, resulting in a drastic decline in snapping turtle populations (Vogt 1981). Some evidence indicates that most of the large turtles are gone from the Mississippi and Lower Wisconsin Rivers (Dan Nedrelo, Wis. Dep. Nat. Resour., pers. comm.). Several long-term snapping turtle trappers have indicated that some local snapping turtle populations have been nearly eliminated by harvest pressures. The demand for softshell turtle meat (two species) has increased significantly for European and Japanese markets. Current regulations do not offer any protection for these species and harvest numbers are unknown except for turtles taken by commercial operations as incidental catch. These records indicate that incidental catch of turtles for the last three years has been the highest in 40 years (Marron 1994).

Modernization of wild rice harvesting methods has led to significant overharvest, resulting in lower yield and elimination of some stands (Bernthal et al. 1992, Vennum 1988), although this problem may have been mitigated by current harvest regulation.

Large-Scale Chemical Treatments

Some water bodies which are thought to have undesirable aquatic communities or high numbers of exotics such as carp are chemically treated and restocked with a desired species mix. Chemical treatments usually eliminate all fish and many other aquatic species in a water body including native species and can have at least regional impacts on biodiversity. Becker (1983), in noting the occurrence of a rare bullhead minnow population in the upper Fox River, stated, "the continued poisoning of portions of the Fox River and adjacent waters with antimycin or other fish toxicants, for the purpose of carp removal, may jeopardize or wipe out the only known Great Lakes population of the bullhead minnow." Chemical treatments of the Rock River system in the 1970s may have eliminated the least darter, a species of special concern from the Maunesha River system (Fago 1992, Becker 1983).

The impacts of chemical treatments are probably confined to the waters treated. Most treated waters are small, but occasionally larger waters such as Beaver Dam Lake, Dodge County (6,542 acres); Delavan Lake, Walworth County (2,072 acres); Horicon Marsh; or the upper Rock River system are treated. Many treated waters were already perturbed by exotics or hyper-eutrophication, and treatments may have improved conditions for native species. On a landscape scale, the relatively small proportion of waters treated make it unlikely that chemical treatments have had a major impact on Wisconsin's biodiversity. However, the lack of a long-term biological monitoring program makes it difficult to determine what effects such treatments have had.

Department Management Priorities

The effects of Department management activities—such as chemical treatments, intensive aquatic habitat alteration, and water level manipulations—on nongame or other nontarget aquatic species is sometimes not considered, given a cursory look, or the nongame species are deemed less desirable than game fish species that have more sport-fishing value. Although these choices are not inherently wrong, they should be made with due consideration to all components of the aquatic ecosystem, particularly on a regional and landscape scale. The need to approach management from an ecosystem management perspective is becoming increasingly evident. Integration of the appropriate programs is essential if we are to respond to the needs of the ecosystem as a whole.

Habitat Improvement Projects

Intensive habitat management practices include: placement of artificial habitat structures such as fish cribs; riprapping or other bank stabilization; construction of channel modification or maintenance structures such as the LUNKER structure (Vetrano 1988); placement of artificially constructed spawning reefs; and spring pond dredging. The impacts of these practices on local biodiversity are unclear. For example, Carline and Brynildson (1977) studied the results of spring pond dredging. Invertebrates usually recolonized dredged areas provided some undisturbed habitat was left. Fish populations sometimes did not change dramatically. Bank stabilization and other channel modification practices were often applied primarily in streams with already disturbed communities with the intent of reestablishing healthy aquatic communities. However, wood turtles typically nest in areas exposed by natural erosion processes along streams (David Evenson, Wis. Dep. Nat. Resour., pers. comm.).

On a statewide scale, it is unlikely that these habitat management practices have had a measurable impact on Wisconsin's biodiversity. Cold-water stream habitat improvement structures have been placed on about 300 miles of Wisconsin's 9,500 miles of trout stream and 33,000 total miles of rivers and streams. The Department has dredged about 60 of the state's 1,700 spring ponds (Carline and Brynildson 1977; Max Johnson, Wis. Dep. Nat. Resour., pers. comm.); some of these were already impacted by agricultural or timber harvest related sedimentation.

Water Level Manipulations

Artificial water level regulation in Wisconsin's 1,550 dammed lakes can have negative impacts on the aquatic system particularly when it deviates substantially from natural patterns. Water levels fluctuate widely in some reservoirs, especially those used for flood control and peaking hydropower operations (Thuemler et al. 1989). Winter drawdowns are frequently done to minimize ice damage to shoreline properties and increase spring runoff storage capacity. These fluctuations have direct impacts when fish or amphibian eggs are stranded and can have indirect impacts when changing water levels favor certain species over others. For example, water level fluctuations apparently favor carp and inhibit reproduction of native northern pike in Petenwell and Castle Rock reservoirs (Jim Kreitlow, Wis. Dep. Nat. Resour., pers. comm.), and low or fluctuating early spring water levels disrupt spawning of northern pike and walleyes (e.g., Johnson 1961, Johnson 1971, McCarraher and Thomas 1972, Holland and Huston 1984, Kallemeyn 1987). Hibernating turtles are susceptible to freezing and desiccation if reservoirs are lowered or drained (Dorff 1990). Heath (1992, 1993*b*, 1993*c*) reported an inverse relationship between the degree of late fall and winter drawdown and turtle population densities in several Wisconsin reservoirs. A species that matures very slowly, like the Blanding's turtle, can be significantly impacted by winter drawdowns, especially since they are already threatened by fragmentation and the loss of habitat (Dorff 1990). Impacts of winter drawdowns on turtles could be minimized by conducting drawdowns prior to October 1st in Wisconsin waters which will allow turtles to seek alternate hibernation sites (Dorff 1990, Heath 1992, 1993*b*, 1993*c*).

In addition, when ice is lowered onto the bottom substrate during a winter drawdown, substrate freezes to the underside of the ice, resulting in scouring and resuspension of sediments (Glenn Miller, Great Lakes Indian Fish and Wildl. Comm., pers. comm.). Organic substrates are both compacted and removed by ice settling on it. During turtle surveys conducted for FERC relicensing on the Chippewa and Peshtigo rivers, it was noted that organic substrates were almost non-existent or compacted in the shallower bays of reservoirs where winter drawdowns were routinely done. Macrophyte plant densities in two reservoirs on the Peshtigo River were significantly less than at other reservoirs on the same river where winter drawdown had not occurred. Correspondingly, turtle populations in these impacted reservoirs were markedly depressed (R. Hay, Wis. Dep. Nat. Resour., unpubl. data).

Water levels are often held higher during summer to facilitate recreational activities and dock access. Higher summer water level may increase littoral habitat for fish (Johnson 1971) or waterfowl, but may increase spawning habitat for carp or affect aquatic macrophytes such as wild rice (Fannucchi et al. 1986). Artificially high water levels in large, shallow lakes can also increase wave damage to littoral areas, increase turbidity, and eliminate macrophytes (e.g., Engel and Nichols 1994). Water levels in large reservoirs are often systematically drawn down to augment summer river flows. Unstable water levels typically result in poor macrophyte development with associated loss of fish and macroinvertebrate habitat, and in extreme cases can impair fish spawning activities. Drawdowns conducted for the purpose of establishing emergent vegetation can increase the opportunity for seeds of exotic nuisance species such as purple loosestrife to germinate and become established (Merendino et al. 1990).

All of the drowned bay mouth estuaries in Lake Superior and many in Lake Michigan are located in Wisconsin. These unique features occur when the mouths of the tributary rivers have formed estuaries enclosed within sand spits formed by along-shore currents, coming and going with changing water levels and storms. These estuaries provide critical habitat for several bird species such as the least tern and piping plover, as well as dune thistle, dwarf lake iris, beach pea, and grass of parnassus.

Estuary Habitat Management

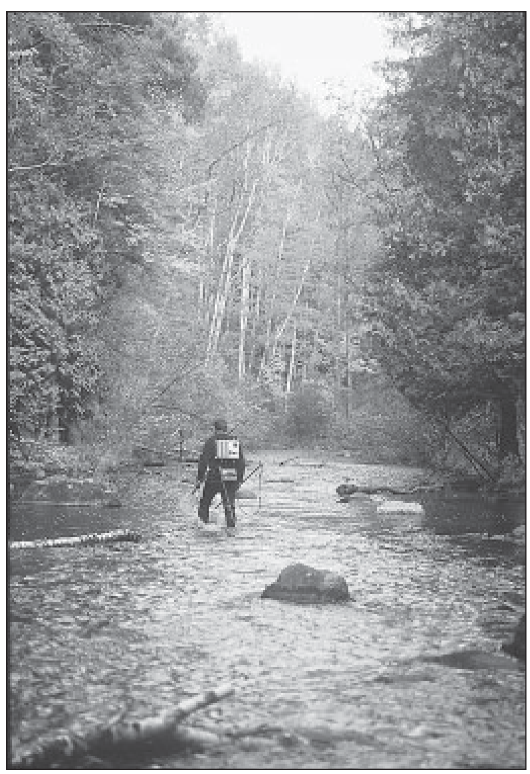
All of the drowned bay mouth estuaries in Lake Superior and many in Lake Michigan are located in Wisconsin. These unique features occur when the mouths of the tributary rivers have formed estuaries enclosed within sand spits formed by along-shore currents, coming and going with changing water levels and storms. These estuaries provide critical habitat for several bird species such as the least tern and piping plover, as well as dune thistle, dwarf lake iris, beach pea, and grass of parnassus.

Many of these estuaries have been degraded by filling and dredging for disposal of industrial wastes, fly ash, taconite tailings, bark and sawdust, and for construction of roads, tank farms, coal storage areas, docks, grain elevators, and small-boat harbors. Developments and alterations are isolating the remaining pockets of estuary (e.g., the separation of St. Louis River estuary from the Allouez Bay estuary), and are causing habitat simplification. Most of Green Bay's Atkinson's Marsh is affected by dredging and filling with fly and bottom ash, dredge spoil, and other wastes behind a bulkhead line. Other estuaries such as at Port Wing, Marinette, and Peshtigo are affected by recreational development, but undisturbed parts also remain. Only a few, such as Flag River and the Kakagon Sloughs, are relatively undisturbed. The Mink River estuary in Door County and some parts of the Kakagon Sloughs have been the focus of protection efforts of The Nature Conservancy.

Lack of Monitoring

The Department has collected a substantial amount of information on the state's aquatic ecosystems over the years. Several statewide surveys of fish distribution have been done (Greene 1935, Becker 1983, Fago 1992). However, these surveys did not provide information on relative abundance of species, and the last survey (in the 1980s) covered only 45% of the state's waters. Other Department or university fish sampling programs cover few waters or a short period of time. The Department's ambient lakes monitoring program collects water quality and limnological information from 50 selected waters but has been in existence for only a decade. The Waters Classification Program, a major statewide survey of physical, limnological, and fishery characteristics of the state's waters, was conducted in the 1960s and 1970s (e.g., Carlson and Andrews 1977). A randomized survey of limnological characteristics of 1,140 lakes was also conducted (Lillie and Mason 1983). The Bureau of Endangered Resources maintains a database of rare aquatic species occurrences, but this is not the result of a systematic inventory. The Bureau of Water Resources Management has also recently initiated a program to collect limnological and macroinvertebrate information as part of their Basin Plan monitoring program. The Department has also conducted many surveys of state waters that were not part of a statewide or regional program.

Numerous surveys of other aquatic organisms have been conducted. Wisconsin's snail populations were surveyed in the 1920s (Baker 1928*a*). Statewide surveys of mussels were conducted in the 1920s and 1970s (Baker 1928*b*, Mathiak 1979), and mussels in the Mississippi River were surveyed in the late 1970s (Ecological Analysts 1981, Theil 1981, Duncan and Thiel 1983). Statewide surveys of crayfish and shrimp were published by Bundy (1882), Creaser (1932), and Hobbs and Jass (1988). Vogt (1981) provided a



Backpack electroshocking equipment allows managers to survey and monitor remote waters. *Photo by Robert Queen.*

summary of collections and descriptions of Wisconsin's amphibians and reptiles, which have been examined in varying degrees of completeness since 1883. The Milwaukee Public Museum maintains a Herpetological Atlas Project which is the current repository for information from these and later surveys. A volunteer frog and toad monitoring survey has been conducted by the Department since 1984 (Mossman and Hine 1984, Mossman and Hine 1985, Mossman and Huff 1990).

Unfortunately, this apparent wealth of survey information falls far short of meeting the critical need for long-term monitoring of Wisconsin's aquatic ecosystems and statewide systematic inventories of aquatic organisms. Data from these surveys are usually not collected in a standardized manner and rarely contain relative abundance information, and surveys are not systematically repeated to track distribution and abundance trends. Information from these surveys is often dispersed among different programs in a variety of computer and paper file storage formats making accessibility difficult. There is a critical need for long-term trend information on the status of aquatic communities using either bioindicator species or community samples. Such trend information will only be obtained by institutionalizing a standardized, statistically valid aquatic ecosystem monitoring program as an integral part of the Department's aquatic management programs. An important basis for such a program would be a systematic inventory of important aquatic organisms across the state's waters.

Bioengineering

Biological engineering to produce faster growing, larger, and more prolific fish species has the potential to alter management goals and objectives from reliance on existing species and strains to designing new species and strains to meet specific management goals and objectives. For instance, it is theoretically possible to engineer more appealing bait fish, new predators, or freshwater fish that continue to grow throughout their lifetimes and attain a weight of several hundred pounds. Past experience with exotic species and predictive management are not accurate or reliable indicators of likely outcomes from adaptive management and biological engineering. However, to the extent that an emphasis and reliance on management diverges from an emphasis on less management with reliance on natural reproduction and a sustainable resource base, conflicts and controversy will result. Judgments on whether the outcomes of these divergent approaches are "good" or "bad" will also depend on the observer's values about resource management.

Beyond these important concerns, there is very little evidence that genetic bioengineering is a viable management option. We barely understand the importance of genetic diversity in native species which have adapted to Wisconsin's waters over millennia. Thus, costly genetic experimentation poses the potential for major disruption of existing aquatic ecosystems.

Recreation

Recreational use of aquatic ecosystems continues to increase. Activities such as boating, canoeing, swimming, camping, and hiking along riparian areas are among Wisconsinites' favored activities (Penaloza 1989, 1991, Wis. Dep. Nat. Resour. 1991). Boats and motors are becoming more numerous and larger (Penaloza 1991). Along with increased demand for these activities comes increasing demand for boat launch ramp, canoe access, beach, marina, campsite, and trail development. Both the recreational use and associated development impact the aquatic ecosystem.

Boating can result in direct impacts to habitat. Outboard motors discharge raw fuel, oils, and other combustion byproducts directly into the water (Jackivicz and Kuzminski 1973, Wall and Wright 1977). Extensive fish kills in the Fox River have been attributed to carbon monoxide discharge from outboard motors (Jim Kempinger, Wis. Dep. Nat. Resour., pers. comm.), though this was an area of exceptionally high use. Heavy boat traffic is known to disturb vegetation and macroinvertebrate production and cause shoreline erosion and sediment resuspension (Lagler et al. 1950, Liddle and Scorgie 1980, Smart et al. 1985). Increased sedimentation is known to have detrimental impacts on fish and other aquatic organisms (e.g., Berkman and Rabeni 1987, Ritchie 1972, Ellis 1936). Heavy boat use is often blamed for turbidity which can also adversely affect aquatic organisms (e.g., Van Oosten 1945, Gardner 1981, Breitburg 1988, Robel 1961, Newcombe and MacDonald 1991, Lloyd et al. 1987,), but there is mixed evidence for this relationship (Lagler et al. 1950, Moss 1977, Liddle and Scorgie 1980, Yousef et al. 1980). Boat traffic also disturbs waterfowl and other aquatic wildlife. (See review by York 1994.) Movement of boats among waters can transport propagules of exotic species such as Eurasian water milfoil and zebra mussels.

Heavy use of riparian areas can result in bank erosion, vegetative destruction, littering, and demands for increased camping and access site development (e.g., Manning 1979). Recent interest in marina construction in the Minnesota-Wisconsin boundary waters of the Mississippi and St. Croix rivers will add to congestion on what is already one of the most crowded and congested areas in the country for boating (Tom Watkins, Wis. Dep. Nat. Resour., pers. comm.). There is some evidence that pollution levels are higher near marinas (Mack and D'Itri 1973). Habitat will also be lost due to dredging and riprapping for development of facilities to support recreational activities such as marinas, condominiums, boat launches, and parks.

While perhaps not of great historical impact on Wisconsin's aquatic ecosystems, the current effects of concentrated boating and other water recreational activities are documented. Given the dramatic increase in the levels of these activities in recent years, resulting changes in the aquatic communities must be monitored carefully.

Recreational boating is a favorite activity in Wisconsin. Boats and motors are becoming more numerous and larger, placing increased pressure on aquatic systems. *Photo by Robert Queen.*

PRESENT STATUS

Quantitative analyses of the present status of aquatic ecosystems in Wisconsin require a statewide database of systematically collected biological samples. Such a database exists only for fish (Greene 1935, Becker 1983, Fago 1992). Other taxa including aquatic macrophytes, phytoplankton, zooplankton, benthic invertebrates, crayfish, amphibians, and some reptile, bird, and mammal species are clearly dependent on aquatic systems, but there is far less systematic data on their distribution and abundance. This section will use fish distribution data to quantitatively assess present aquatic ecosystem health across the state.

The status of the aquatic communities can be successfully indexed by fish communities (Karr 1981, Fausch et al. 1990, Lyons 1992). Extensive fish surveys of Wisconsin waters by Greene (1935), Becker

Aquatic macrophytes are an important component of aquatic biodiversity. Systematic data on their distribution and abundance is needed. *Photo by Dorothy Cassoday.*

(1983) and Fago (1992) provide a quantitative basis for discussion of the current status of Wisconsin's aquatic communities. Use of indicators such as presence of fish species intolerant to environmental degradation (Lyons 1992), species richness, history of extirpations, current status of threatened species, and status of natural reproduction of top-level predators show trends in aquatic ecosystem health.

Great Lakes

The shoreline of parts of the Great Lakes has been modified by urban, industrial, and second-home development, especially in urban areas such as Duluth-Superior, Green Bay, Milwaukee and along the Lake Michigan shore near the Illinois state line. Shoreline protection efforts such as groins, jetties, cribs, dredging, and navigation channel entries have interfered with long shore movement of littoral drift. Erosion and scouring of the down-drift side of these structures is occurring. Other shoreline changes such as riprap, sheet pile walls, gabions, or concrete retaining walls are used in an attempt to stabilize the shoreline. They retard the natural process of beach formation and destroy unique beach and bank plant communities. Alteration of water levels and natural fluctuations has affected dune and coastal marsh systems by interrupting nutrient and organic matter flushing (The Nature Conservancy 1994). Changing water level fluctuation cycles is leading to the simplification of coastal marshes by eliminating species that require drawdowns at certain times to allow germination (Keddy and Reznicek 1986).

Contaminants, particularly PCBs, are commonly found in many Lake Michigan and Green Bay fish and waterfowl at levels which require consumption advisories. Contaminants in the Great Lakes sediments and waters have been passed along through the food chain, resulting in contamination of invertebrates, fish, wildlife, and humans. Contaminants remain in the bottom sediments, and fish in remote parts of Lake Superior have mercury and PCB in their tissues, indicating contamination may still be occurring due to atmospheric deposition. Monitoring indicates that contaminant levels have been declining in Lake Michigan fish since the 1970s (e.g., Staggs 1987) though rates of decline have slowed because of internal cycling and continued atmospheric deposition (Arimoto 1989). Monitoring of Lake Superior fish shows lower contaminant levels.

The Great Lakes aquatic community has also been affected by humans. The Great Lakes fish community is a good example. The effects of lamprey predation, water quality degradation, invasion of exotics, and overfishing have combined to radically alter the fishery. Some species of cisco have been extirpated in Lake Michigan. Strains of other species, such as lake trout and brook trout, have been eliminated or greatly reduced. Introductions and invasions of non-native fish species, such as rainbow and brown trout, Pacific salmon, smelt, and alewife have changed the species composition of the fishery. However, the introduction of the salmon resulted in the decline of alewives and a comeback in native perch, sculpins, and coregonids (Stewart et al. 1981, Eck and Brown 1985, Jude and Tesar 1985, Wells and Hatch 1985). Other species, such as perch, undergo periods of intensive harvest. Undoubtedly predator/prey relationships have been affected by these changes in the fish community.

The status of the fish communities in Lake Michigan is best described as disturbed and unstable (Wells and McLain 1973). Reproduction of trout and salmon is negligible, and populations are primarily supported by stocking. Sea lamprey predation, angler harvest (Clark and Huang 1985), and overstocking (Stewart et al. 1981) all affect fish populations. Reproduction and populations of native deepwater sculpins and bloaters are at recent highs, while non-native alewive populations are extremely variable but are at recent lows (Jude and Tesar 1985, Wells and Hatch 1985). Populations of lake and round whitefish seem abundant and relatively stable (Wells and McLain 1973). Of the seven cisco species once found in Lake Michigan, only the cisco and bloater remain in numbers sufficient to preserve the population. Three cisco species—shortnose, blackfin and deepwater—are likely extinct (Becker 1983). Two other cisco species—shortjaw and kiyi—while likely extirpated in Lake Michigan are still relatively abundant in Lake Superior. Excessive commercial harvest and competition from alewives are cited as primary causes of cisco population declines.

Among Lake Michigan warm-water species, reproduction and populations of northern pike are currently limited. Walleye and yellow perch reproduction is now significant in Green Bay, but populations are still experiencing large fluctuations. Yellow perch are well-established in other areas such as Milwaukee harbor, but walleye populations are negligible. Lake sturgeon populations are probably low but the current level of reproduction of this long-lived species is unknown. Reproduction of other warm-water species appears adequate to maintain the stocks.

In Lake Superior, fish communities, although heavily exploited, are more stable (e.g., Lawrie and Rahrer 1973). There is significant natural reproduction of most trout and salmon species, but angler and commercial harvest and sea lamprey predation have kept adult populations at a relatively modest level. Reproduction and populations of other cold-water species is adequate, but overall productivity in Lake Superior is low so populations are often modest by Lake Michigan standards and cannot support as large a predator population. Only four cisco species—cisco, bloater, shortjaw, and kiyi—were originally present in Lake Superior and all are still present in sufficient numbers to maintain populations. Lake Superior is also home to the only known population of pygmy whitefish east of the Rockies. It is abundant and relatively stable (Becker 1983). With some local exceptions, reproduction and populations of Lake Superior warm-water species are adequate to maintain the stocks.

Fish communities—specifically the abundance of lake trout—can be used as indicators of the status of the entire aquatic community in the Great Lakes (Ryder and Edwards 1985, Marshall et al. 1987). Based on this assumption, Lake Michigan has been dramatically affected by habitat simplification— primarily dredging, wetland filling and water quality declines in estuarine areas, introduction of exotic species, excessive harvest of commercially desirable top predators, and pollution.

The uncertain status of lake sturgeon reproduction indicates fragmentation of the lake ecosystem, as lake populations were cut off from historical spawning areas by dam construction. Lake Michigan, however, shows some signs of recovery: declining numbers of exotic species, improving reproduction of some native species, and declining contaminant burdens (e.g., Wells and McLain 1973).

Conversely, Lake Superior shows few signs of either habitat simplification or fragmentation. The aquatic community has primarily been affected by human management activities, including excessive harvest of commercially desirable species, stocking of domesticated strains of lake trout, and introduction of exotic species (Lawrie and Rahrer 1973). Localized habitat degradation in the urban areas of Duluth-Superior may also be occurring but does not appear to be affecting Lake Superior biodiversity on a lake-wide scale.

Inland Lakes

Fish species intolerant of poor water quality and environmental degradation (Lyons 1992) such as smallmouth bass, rock bass, Iowa darter, blacknose shiner, and spottail shiner were found at 56% of the 1,644 sampled lake stations (Table 19). Species richness averaged 7.2 species per station with a range of one to 23 species (Table 20).

There are currently no federally threatened or endangered fish species in Wisconsin lakes, although six species are under consideration for federal listing. No known extinct species were endemic to Wisconsin lakes (Becker 1983). Several species are thought to have been extirpated

Table 19

Percent of stations with tolerant and intolerant fish species in Wisconsin lakes and rivers, classified by ecoregion as defined by Omernik and Gallant (1988), based on nearest county boundary. (Data source: Wisconsin Fish Distribution Study Master Fish File.)

Ecoregion	Total Stations	Stations with Intolerant Species*	Percent with Intolerant Species*	Stations with Tolerant Species*	Percent with Tolerant Species*
Lakes					
Driftless Area (DRFT)	22	11	50	17	77
N. C. Hardwood Forest (NCHF)	355	203	57	266	75
N. Lakes and Forest (NOLF)	660	424	64	511	77
S.E. Wis. Till Plains (SETP)	607	276	45	471	78
Statewide	1,644	914	56	1,265	77
Streams					
Driftless Area (DRFT)	1,586	886	56	1,466	92
N. C. Hardwood Forest (NCHF)	1,079	850	79	977	91
N. Lakes and Forest (NOLF)	1,317	1,029	78	1,149	87
S.E. Wis. Till Plains (SETP)	1,433	662	46	1,376	96
Statewide	5,415	3,427	63	4,968	92
Rivers					
Driftless Area (DRFT)	331	289	87	245	74
N. C. Hardwood Forest (NCHF)	75	46	61	55	73
N. Lakes and Forest (NOLF)	68	59	87	39	57
S.E. Wis. Till Plains (SETP)	26	19	73	20	77
Statewide	500	413	83	359	72

*Tolerance and intolerance to environmental degradation as defined by Lyons (1992).

from Wisconsin waters, including ghost shiner, ironcolor shiner, and creek chubsucker, but these were probably not common in lakes. Wisconsin lists nine species of endangered fish and 11 fish species as threatened, although most of these species are on the edge of their range in Wisconsin and were never common in lakes.

Differences in fish communities among ecoregions suggest that biodiversity in SETP lakes has been more heavily influenced by human activities compared with NOLF and NCHF lakes. Comparisons

Table 20

Analysis of fish species richness in Wisconsin lakes and rivers, classified by ecoregion as defined by Omernik and Gallant (1988), based on nearest county boundary. (Data source: Wisconsin Fish Distribution Study Master Fish File.)

Ecoregion	No. Fish Species, by Water Type							
	Lake		River		Stream		All	
	Mean	STD	Mean	STD	Mean	STD	Mean	STD
Driftless Area (DRFT)	8.00	4.15	15.19	7.36	9.08	5.43	10.11	6.23
N. C. Hardwood Forest (NCHF)	7.41	4.02	11.53	7.99	10.35	6.30	9.72	6.08
Lakes and Forest (NOLF)	7.02	3.52	11.99	5.92	8.10	4.59	7.88	4.42
S. E. Wis. Till Plains (SETP)	7.19	4.29	11.42	8.76	9.88	5.76	9.11	5.56
All	7.18	3.94	14.01	7.52	9.31	5.58	9.16	5.64

with DRFT lakes are difficult because few lakes exist or were sampled in that ecoregion. Species regarded as intolerant of environmental degradation were present at only 45% of sampled SETP stations (Table 19). Five state-threatened and one state-endangered species were found in SETP lakes including pugnose shiner, redfin shiner, river redhorse, starhead topminnow, striped shiner, and longear sunfish.

In contrast, fish communities in NOLF and NCHF lakes showed less evidence of biodiversity impacts. Species intolerant of environmental degradation were present at 64% of sampled NOLF stations and 57% of NCHF stations. Tolerant species such as carp and green sunfish are generally uncommon (see Table 13). Intolerant species such as the small-mouth bass, rock bass, and Iowa darter are more common. Only the state threatened pugnose shiner, redfin shiner, Ozark minnow, and longear sunfish were found in any NOLF or NCHF lakes, and it is unlikely that any species were extirpated from lakes in these regions. One species thought to be extirpated from SETP waters (Becker 1983), the black redhorse, was recently found in a NCHF reservoir (Fago and Hauber 1993).

The current status of NOLF and NCHF lakes as indexed by fish communities is healthy and stable. Reproduction and abundance of top level predators is generally adequate (Staggs et al. 1990, U.S. Dep. Int. 1991), and a large proportion of sampled stations have species that are intolerant to environmental degradation. Species richness is less than that of southern Wisconsin lakes, but waters in more northerly latitudes are often species poor to begin with (Lyons 1992).

Localized impacts such as dam construction in NCHF lakes are not readily apparent in this regional analysis but are known to be important in specific waters. Water level fluctuations and high nutrient loading are thought to have resulted in poor water quality and high carp populations in the region's two largest reservoirs, Castle Rock and Petenwell (Jim Kreitlow, Wis. Dep. Nat. Resour., pers. comm.).

Piscivorous birds have been impacted by water quality problems in NOLF lakes. Common loons rarely nest on lakes with low pH and elevated mercury levels and suffer higher reproductive mortality in these locations (Meyer 1994).

With regard to aquatic vegetation, wild rice areas in northern Wisconsin have been lost to flooding caused by dams (Vennum 1988).

Streams

No federally threatened or endangered fish species are found in Wisconsin streams, but six species are currently under consideration for federal listing. At least three species, ghost shiner, ironcolor shiner and creek chubsucker, have been extirpated from state streams. The nine state-endangered and 11 state-threatened species are found in state streams. Most are on the edge of their distribution, but species such as the river redhorse, pallid shiner, crystal darter, and gilt darter are declining across their ranges. Wisconsin has some of the best populations of greater redhorse and pugnose shiners across their ranges (Lee et al. 1980) even though they are listed as threatened in this state's streams.

Species intolerant of environmental degradation, such as brook trout, small-mouth bass, and rock bass (Lyons 1992), were found at 63% of the 5,415 stations sampled statewide (Table 19). Species richness is higher in streams than lakes, averaging 9.3 statewide and with a range of one to 40 species per station (Table 20).

In comparison with NOLF and NLHF streams, patterns of fish distribution suggest that some streams in the SETP and DRFT ecoregions have diminished biological integrity. Intolerant species were found at only 46% and 56%, respectively, of sampled stations. While many species shifts are due to underlying habitat differences such as larger streams and warmer temperatures, the increased abundance of tolerant species such as carp, green sunfish, bluntnose and fathead minnows, and yellow bullhead provide evidence for environmental perturbation (Table 15).

NOLF	Northern Lakes and Forest
NCHF	North Central Hardwood Forest
DRFT	Driftless Area
SETP	Southeast Wisconsin Till Plains

Other studies suggest that smallmouth bass populations in DRFT streams have experienced major declines during the last three decades (Forbes 1985, Mason et al. 1991).

Two state-endangered and eight state-threatened fish species were found at SETP stations; one state-endangered and nine state-threatened fish species were found at DFRT stations; no state-threatened fish species were found at NCHF stations; and three state-threatened fish species were found at NOLF stations.

Large Rivers

Species regarded as intolerant to environmental degradation (Lyons 1992) occurred at 83% of sampled stations. Rivers exhibit the highest species richness of all the aquatic communities, averaging 14 species with a range of one to 40 species per station.

No federally threatened or endangered fish species are found in Wisconsin rivers, but six species are currently under consideration for federal listing. At least three species, ghost shiner, ironcolor shiner and creek chubsucker, have been extirpated from state streams and rivers. The nine state-endangered fish species and many of the 11 state-threatened fish species currently inhabit state rivers. Most are on the edge of their distribution, but species such as the paddlefish, blue sucker, river redhorse, pallid shiner, crystal darter, and gilt darter are declining across their ranges. Wisconsin has some of the best populations of greater redhorse across their ranges (Lee et al. 1980) even though they are listed as threatened in this state's rivers.

Differences in river fish communities between ecoregions were not pronounced. Average species richness is higher in the DRFT ecoregion, but this is probably because the largest rivers, the Mississippi River and the lower Wisconsin River, are located here. Species intolerant to environmental degradation are found at a large percentage of sampling sites in all ecoregions (Table 17). Intolerant species were found at only 61% of NCHF river stations perhaps reflecting impacts of damming and paper mill pollution on the Wisconsin and Chippewa rivers.

Several state-threatened fish species are common in NOLF, NCHF, and DRFT rivers. Only three state-threatened species—gilt darter, river redhorse, and blue sucker—have been found in NOLF rivers, but the gilt darter and river redhorse were found at one third of the sampled stations suggesting they are not threatened in the NOLF ecoregion. Seven state-threatened fish species were found in NCHF rivers. DRFT rivers have nine state-threatened and one state-endangered fish species. The four state-threatened and one state-endangered fish species found in SETP rivers but are not common in that region's rivers.

NOLF rivers contain the lowest percentage of intolerant species among the four ecoregions suggesting there has been environmental degradation in the ecoregion. However, there is evidence that NOLF fish communities are still relatively healthy. River redhorse, a state-threatened species known to be affected by dam construction, was present at 30% of sampled stations. Also, the gilt darter, a state-threatened species intolerant of environmental degradation, was found at 32% of sampled stations. Most top predator species exhibit self-sustaining populations.

These analyses show that some stations have degraded fish communities, while fish communities at many stations remain intact. This finding is consistent with the observation that dam construction is a major environmental impact on Wisconsin's rivers since the most significant effects of a dam would be localized in areas near the dam.

Projected Status

Some taxa dependent on aquatic systems in Wisconsin face a difficult future. The abundance and zoogeographical distribution of Wisconsin's river mussels have been dramatically altered. Three species have been extirpated (scaleshell, fat pocketbook and pyramid pigtoe), two have only remnant populations (ebony shell and elephant ear), and two are on the federal endangered species list (winged mapleleaf, higgins eye). If the invading zebra mussel affects native mussel species as severely as predicted, up to half of the states stream and river mussel species will be threatened or endangered.

Other taxa are in better condition. Although the abundance of many fish species has been greatly altered, very few fish species have been extirpated in Wisconsin. Of the fish species locally extirpated, some have later been found again, although in very low numbers (e.g., black redhorse and skipjack herring), and most extirpations involved species on the edge of their range. There are no federally endangered or threatened fish species in Wisconsin, although the lake sturgeon, blue sucker and paddlefish are among the candidates for listing. It appears most extirpations have been caused by habitat destruction, primarily associated with agriculture and dams. In general, most native fish species are self-sustaining, especially in Lake Superior and the northern ecoregions. Even the heavily developed agricultural areas and densely populated areas of southeastern Wisconsin still support self-sustaining fish populations and good species diversity in some waters.

The relative abundance of currently healthy aquatic communities in many inland waters is an excellent indicator for the future. Attention can be focused on identification and restoration of specific degraded habitats and on protecting and restoring species whose numbers are in local decline. Existing aquatic communities provide a source for recolonization of native species and a model for restoration efforts. Past restoration successes, such as those on the Wisconsin and lower Fox Rivers, provide clear evidence that such efforts will work. Further progress and additional successes can be expected as the Department works cooperatively with industry to install the latest pollution abatement equipment and develop pollution prevention technology. Many priority watershed plans already recommend removal of dams to improve water quality. Dam removal may be an effective option for restoration of riverine ecosystems.

The most cost-effective management strategy is protection of existing healthy, self-sustaining aquatic ecosystems. However, pressure remains to develop and destroy riparian habitat, modify land use patterns in watersheds, intensify agriculture, divert water for irrigation and industrial uses, and provide more harvest opportunities for sport anglers and subsistence users. Government agencies and users will have to establish and maintain strong partnerships if long-term sustainability of aquatic ecosystems is to be maintained.

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The accidental or intentional introduction of exotics has already been implicated in major changes in native biodiversity. Past invaders, including carp, Eurasian milfoil, and purple loosestrife are notoriously hard to control and eradicate. The state has invested heavily in control, education, and monitoring, but efforts to eliminate targeted exotics have shown poor results. Undoubtedly, new exotic species will continue to be introduced, and the potential threats to the aquatic community will increase.

The abundance and distribution of Wisconsin's river mussels have been dramatically altered. The winged mapleleaf, shown here, is on the state and federal endangered species lists. *Photo by William Smith.*

Healthy, diverse aquatic communities are present in many inland waters. *Photo by Dean Tvedt.*

Two other possible threats to the aquatic community present unknown dangers—the potential effects of climate change and the effects of acid deposition (Bergquist 1991). Assessing the likelihood of either of these threats is dependent on the development of accurate predictive models. Global climate change could change the numbers and distribution of aquatic species in Wisconsin. However, the climate change threat is only in the early stages of monitoring and model development (U.S. Dep. Energy 1990).

Socio-Economic Issues

The aquatic community has a faithful and dedicated following of avid anglers, recreationists, and other user groups. Conflicts can occur as more people try to use a limited resource for purposes that are not easily compatible, such as water-skiing and fishing. In some cases, there is a strong feeling of "my lake," "my river," "my trout stream," and "my spot," particularly among local residents and riparian owners, which may place them in conflict with other statewide users.

Recreational fishing has a significant impact on aquatic communities. In Wisconsin, fishing is the sixth most popular outdoor activity among all adults (26% of all adults) (Wis. Dep. Nat. Resour. 1991). In some areas of the country, fishing is now so intense that previously underexploited recreational fisheries are now thought to be approaching overexploitation, although this concern is not well documented in Wisconsin.

The popularity of fishing has come at a price. The state's fishing public, which is among the five largest in the nation (U.S. Dep. Int./U.S. Dep. Comm. 1993), demands an intensive fishery management effort. Resort owners want more large predators such as walleye, muskellunge, and northern pike, while trout anglers want more trout. More access is required to meet the demands of more people who own more boats with bigger motors and who purchase more licenses and pay higher fees. More demands require stretching the resources of the aquatic community to provide for more return, much as a farmer may try to increase yield from a corn field.

The short-term expectations of resort owners and other interests lead to increased pressure for management actions, such as fish-stocking, habitat manipulation, and single species management, rather than practiced restraint and reliance on natural recruitment to replenish a fishery. As the pressure for providing short-term solutions increases, the interest in the long-term health and diversity of the aquatic community could diminish.

There is heavy economic pressure to continue developing shoreline property. Lake homes continue to be in high demand. Some counties, such as Vilas and Oneida, are growing rapidly due to the demand of retirees for lake and waterfront homes. Former resorts are being converted to condominiums. This demand now threatens the smaller, more isolated, shallow lakes that were not developed earlier because they were "less desirable."

The agriculture industry is an extremely important component of Wisconsin's economy, and its impacts on aquatic ecosystems in Wisconsin are well documented. The Priority Watershed program along with stream bank protection acquisitions and easements under the Stewardship Program are the main management activities targeted at reducing this large source of pollutants and erosion, but both are largely voluntary programs.

An evaluation of the socio-economic implications of managing Wisconsin's aquatic ecosystems reveals many of the same problems that mark similar analyses of natural resource issues. Documentation of the value of development and industry needs is straightforward and readily available. Hydropower will generate electricity worth a certain amount. Lakeshore development yields a certain amount of property taxes for local government. Industries using aquatic resources employ a certain number of people and contribute a certain amount of tax revenues to local governments. Family farms employ a certain number of people and generate a certain level of expenditures in the rural communities. However, documentation of the value of the aquatic resource degraded or lost is less certain and often involves hard-to-quantify intrinsic values. What is the value of the nongame species killed by a pollution discharge? What is the value to the local economy when tourism declines because of poor fishing, increased pollution, or loss of scenic beauty? What is it worth to be sure future generations will be able to enjoy clean water, good fishing, and scenic recreational areas? A major challenge to maintaining the long-term sustainability of Wisconsin's aquatic ecosystems will be to develop adequate valuations of the importance and uses of these ecosystems, and to ensure that society and future generations are not paying for short-term benefits that limit future options. However, according to Clark (1991), "the political realities are that exploiters of large resource stocks have every incentive to impose major external costs on the public at large, and these externalized costs add up to nonsustainability."

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Potential for Community Restoration

Aquatic systems can probably recover more quickly than terrestrial systems from the impacts of fragmentation or simplification if the causes are corrected. In rivers and streams, sediments and contaminants are flushed downstream, hydrologic processes will restore channel morphology, riparian areas will revegetate, and aquatic organisms will return provided there are no barriers to recolonization from unaffected areas (Detenbeck et al. 1992). Macroinvertebrates, for example, generally recover very quickly (months to just a few years) from most kinds of disturbances (Niemi et al. 1990) and colonize new habitats very rapidly (e.g., Williams and Hynes 1977, Doeg et al. 1989). Narf (1985) found aquatic insect colonization of available habitat in a relocated stream segment in northern Wisconsin was complete after 5.5 years. Many other aquatic organisms are mobile and fecund, allowing rapid recolonization and repopulation of affected areas.

Restoration of lakes may take longer or require more directed management actions. Lakes that have been exposed to contamination or excessive nutrient loading usually take longer to recover than rivers and streams because flushing rates are much longer and nutrients and contaminants are continually recycled from sediments. Morphology in lakes is not shaped by strong currents, so restoration of altered habitat may take extremely long periods. Revegetation of riparian areas would be similar to moving water systems, but replacement of woody debris habitat would be slower since it is not being actively transported from upstream areas. Lakes also tend to be more isolated from other

Remote lakes such as Gobler Lake in Oneida County offer valuable insight into restoring aquatic communities. This lake is located within a State Natural Area. *Photo by William Tans.*

lakes, which would slow recolonization by aquatic organisms.

There are few if any undisturbed aquatic ecosystems in Wisconsin to use as templates for restoration efforts. However, there are many systems that, while disturbed, still maintain a healthy complement of native species. These systems must serve as models for restoration and as sources of genetic stock for recolonization efforts. When restoration of pre-settlement aquatic ecosystems is a desirable goal, studies of undisturbed systems in other regions or use of paleolimnology techniques may be needed to establish realistic goals. Determining restoration objectives will not always be straightforward. Historically and currently, recreational and commercial fishing demands guide management efforts and some components of the aquatic community may consequently be considered less important in restoration projects. Careful consideration of the costs of different management activities and a balancing of management objectives across various scales must be part of any restoration plan.

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A basic barrier to both restoration and maintenance of sustainable aquatic ecosystems is the lack of meaningful ability to regulate habitat destruction. The Department has only minimal authority to regulate environmental impacts from agriculture. On paper the Department should be able to minimize destruction of riparian habitat areas, but in practice such destruction continues. Land use practices in nonriparian areas of a watershed can have major impacts on downstream aquatic systems, but zoning and other land use regulations rarely consider aquatic impacts.

The successes of point-source water pollution regulation should be a model for regulation of nonpoint pollution and riparian development. The substantial investments made by both governmental agencies and state industries in water quality improvements are showing results and should serve as a model for other restoration efforts. For example, water quality in the Wisconsin and Fox rivers has been considerably improved and the aquatic resource, most notably fish species, has improved as well.

However, many waters in Wisconsin still need attention. The atmospheric transport of pollutants across the continent and throughout the region results in the deposition of combustion byproducts, sulfur and nitrous oxides, mercury, and PCB's from industries in other states or nations to Wisconsin's waters. The results, such as acid deposition, have been implicated in raising the level of mercury contamination in fish from waters that are not exposed to other sources of pollution (Lathrop et al. 1990). Contaminant advisories still remain for some species of fish, such as carp and white bass. In addition, the allowable residue in fish flesh continues to be lowered, reflecting the improving technology of contaminant detection and the interests of Americans in maintaining a healthful lifestyle. Despite some declines in the levels of PCB's in Great Lakes fish (Staggs 1987), the problem of toxics from some materials, such as dioxins and PCB's, will continue for a long time due to their persistence and cycling in the environment, which may be on the order of several hundred years. Some in-place pollutants have been covered by recent deposition of cleaner sediments, but these overlaying layers can be removed during dredging or scouring during floods. Additional sources of pollutants may occur as a result of groundwater inputs from the thousands of abandoned landfills and leaking underground storage tanks.

State industries have already made substantial investments in pollution abatement equipment, which has reduced the level of pollutants being added to aquatic systems. However, remedial actions to clean up past pollution will be technically difficult and very costly. There is increasing recognition among government agencies and state industries that development of pollution prevention technology is a more cost-effective option than paying for later clean-up of pollutants.

There are few selective controls for exotic species and none are cost-effective on a large scale. One method to control exotic fish in small waters is to eradicate the entire fish community and start over. More often than not, the inconvenience of an exotic fish species is tolerated because of the costs, controversy, and difficulty of completely removing it along with most of the rest of the aquatic community. Maintaining a large predator biomass may be helpful over a long period (Stewart et al. 1981), but extirpation of local forage fish species or shifts in species composition of zooplankton and phytoplankton may occur. Herbicides can be applied to small infestations of certain plants, but large scale control is expensive and damaging to related native species. No control methods exist for exotic bivalves and zooplankton. There is evidence that healthy native aquatic ecosystems are more resistant to invasions of exotics (Baltz and Moyle 1993).

Concerns about the effect of climate change on the aquatic community are starting to surface. Regier et al. (1990) describe three levels of connections between climate change and fish. First, there is a direct connection between local climate and local assemblages of fish. Second, there are indirect linkages between climate, hydrology, and the biotic system. Finally, there is the human and cultural response to these changes.

Weather extremes of the past decade and the possibility of global climate change make it difficult to predict any resulting changes in the aquatic community. The past decade has been marked by unprecedented droughts preceded by periods of extreme wetness. The cold-water resources of the northeastern and southwestern parts of the state have been affected by past droughts, and the Great Lakes littoral areas and shorelines have been affected by high water and resulting shoreline erosion and wave damage. The recent variations in climate fall within the bounds of predictions from the atmospheric general circulation model that predicts climate change.

Effects of climate change could include decreases in winter ice cover on the Great Lakes (Sanderson 1987); the development of a permanent thermocline in the deeper parts of Lake Michigan overlain by a seasonal thermocline such as occurs in most of the world's oceans (McCormick 1990); hypolimnetic anoxia (Schertzer and Sawchuk 1990); increased bacterial activity in the hypolimnion and sediments (Blumberg and Di Toro 1990); changes in habitat for Great Lakes cold-, cool-, and warm-water fish (Magnuson et al. 1990); changes in fish growth rates (Hill and Magnuson 1990); reduced stream habitat for brook trout (Meisner 1990); expansion of the range of the exotic white perch in the Great Lakes (Johnson and Evans 1990); extension of the northernmost ranges of yellow perch and smallmouth bass (Shuter and Post 1990); possible local extinctions of southern fish populations and northward invasion of southern fish populations (Tonn 1990); and the invasion of species adapted to warm conditions concurrent with local extinctions of some cold-water species (Mandrak 1989).

Most of the consequences of today's aquatic habitat problems are the result of changes brought about by agriculture, forestry, and urban development practices several decades ago. Some positive trends are now on the horizon, such as the Conservation Reserve Program provisions of the 1985 Food Security Act. In 1988, Wisconsin adopted Water Quality Standards for a broad range of contaminants. The 1990 Farm Bill strengthened the requirements for environmental protection and attempted to lessen the water quality impacts of agriculture (Pajak 1991). However, later versions could change these gains. Additionally, the Department's Stewardship and Forestry Best Management Practice Programs have a substantial component devoted to water resources protection. Increased interest in wetland protection and sustainable agriculture may lead to lower chemical inputs and less erosion, while changes in manufacturing methods and waste treatment portend possible decreases in wastewater discharges.

Possible Actions

The following possible actions are consistent with ecosystem management, but require more analysis and discussion. How priorities are set within this list will be based on ecoregion goals, staff workload, fiscal resources, public input and support, and legal authority. We will work with our customers and clients to set priorities and bring recommendations to the Natural Resources Board for consideration beginning in the 1995-97 biennium.

1. Apply the principles of ecosystem management to the many kinds of aquatic communities and their associated species. Put less emphasis on single species management and more on communities and ecosystems. One benefit will be the cost-effectiveness of managing for native, naturally reproducing species assemblages and up-front protection of habitat.

2. Use a landscape scale approach to set watershed or ecoregion-based goals for the protection and management of lake systems. This will involve work with many partners, public and private, to apply the ecological, socio-economic, and institutional aspects of ecosystem management to a comprehensive view of our lake resources and the conservation of statewide biological diversity.

3. Recognize the importance of protecting certain unique types of aquatic communities (e.g., historic wild rice stands), undisturbed aquatic communities (e.g., wilderness lakes), and long-lived native species (e.g., 100 year-old-ebony shell mussels, 75-year-old snapping turtles, and 120-year-old lake sturgeons). These are often are economically valuable, add stability to ecosystems, are a reservoir for genetic diversity, and have tremendous scientific value for understanding the processes that affect managed and harvested systems. The concept of old growth, usually applied to forest communities, may help us manage and value populations of long-lived aquatic species and species assemblages.

4. Manage rivers as ecological continuums from headwater to mouth, taking into account adjacent floodplain and terrestrial habitats. Recognize the role of floods in maintaining the integrity of river ecosystems. To do this, we will work with many public and private partners to develop ecoregion or watershed goals and objectives based on ecosystem management principles. This will include reaching a consensus on the desired outcomes after considering a full range of management opportunities (e.g., nonpoint source control, recreational use, industrial activity, aquatic community restoration, and enhancement of fisheries and aquatic life).

    a. Emphasize the protection of the last large river systems without dams. These include the Lower Chippewa

River, Lower Black River, and the Namekagon/St. Croix Rivers.

b. Where appropriate, identify opportunities for upstream and downstream fish passage at existing and proposed dams.

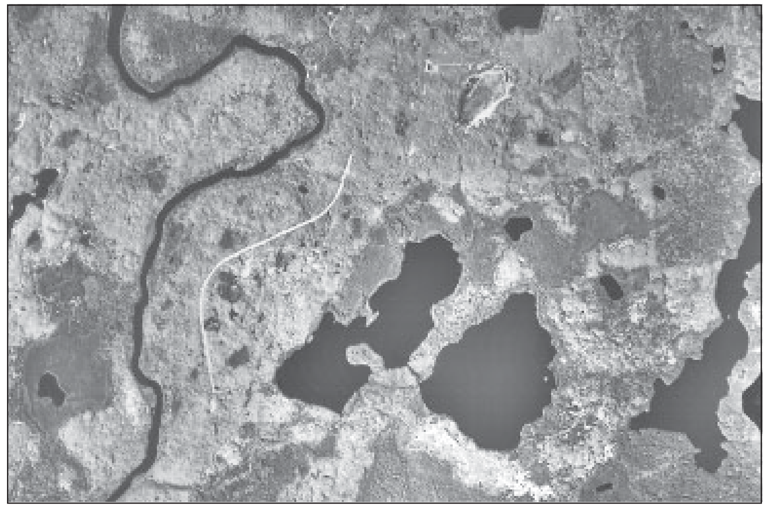
c. Where flows are adversely affected by dams, seek to establish adequate minimum flows to protect recreation, water quality, and fish and aquatic life. Prepare drought contingency plans for rivers where there are consumptive uses or conflicting uses.

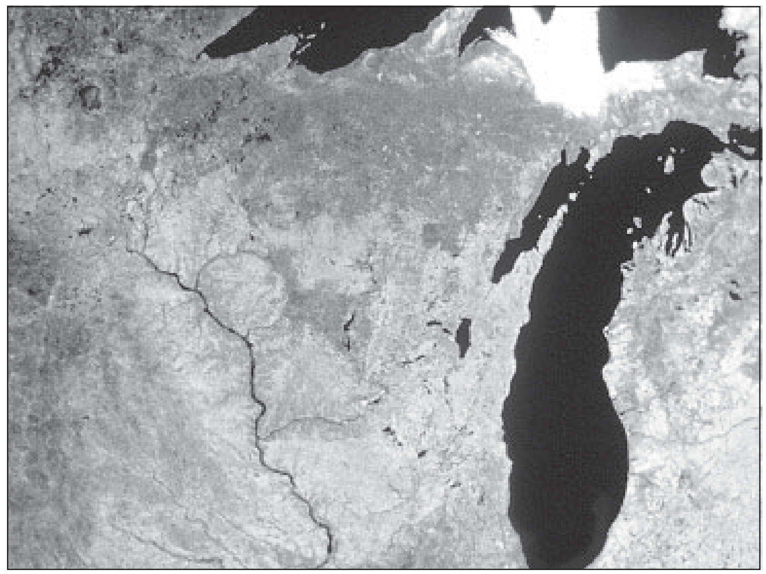
d. Document the cost and benefit of dam removal for selected restoration projects, and use the analysis within the ecosystem management decision model to recommend appropriate action.

e. Encourage the preparation of hydroelectric flow models based on entire river systems.

f. Examine the practice of removing natural woody debris from stream beds for channel maintenance. If needed, prepare guidelines to protect instream habitat structure.

5. Manage riparian and shoreline forests using ecosystem management principles. Allow floodplains to develop mature forests to minimize the impacts of flooding and to maintain channel geomorphology. Use Wisconsin's Forestry Best Management Practices guidelines, which require a buffer area of at least 100 feet along shorelines, to plan timber harvest. These buffers are sources of fallen woody debris to maintain instream habitat structure, and they provide shade to control stream temperature. They also protect banks and ground vegetation from damage caused by heavy equipment.





Aerial photographs and satellite imagery are among the tools that help managers take a landscape scale view of aquatic systems.

[Top] *Photo by National Aerial Photography Program.* **[Bottom]** *Photo by Univ. of Wisconsin-Madison Environmental Remote Sensing Center*

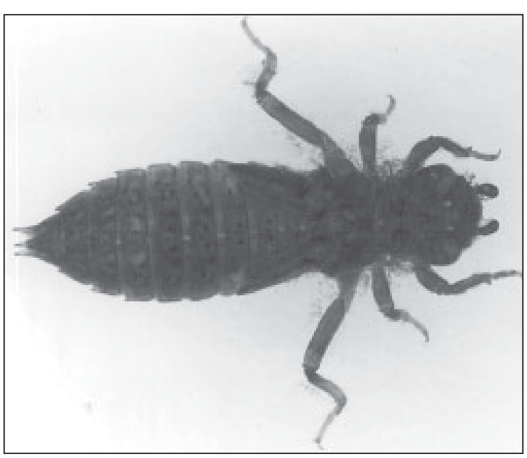
6. Develop programs, regulations, and guidelines that effectively protect riparian and shoreline habitats. This will include work with local governments and private groups to develop a common understanding of the impacts and long-term costs of poorly planned riparian and shoreline development and to provide the support needed to design and implement long-range plans that provide adequate protection. We will need to promote a combination of approaches that include:

- zoning practices, such as those that protect sensitive areas from overuse, disturbance, or destruction (e.g., special designations for spawning areas or undisturbed natural communities);

- alternative methods of protection (e.g., new technologies for erosion control and shoreline protection, incentives for property owners to protect habitats); and

- traditional policies and regulations (e.g., enforcement, legislation, and grants).

7. Identify and restore degraded aquatic communities, working in partnership with other public and private groups to ensure success of these projects. Removal of the Woolen Mills dam and restoration of river and riparian habitat on the Milwaukee River provides an excellent template for similar projects and demonstrates that there is wide support for such activities, particularly in urban areas. Some northern rivers affected by historical log drives are among the candidates for restoration. In some projects, dredging and disposal of contaminated sediments may be necessary, and the lack of an approved hazardous waste site in Wisconsin may present problems. There is great potential for additional joint restoration projects involving state, federal, county, municipal, industrial, and citizen partners. For example, the Great Lakes Indian Fish and Wildlife Commission is interested in restoring wild rice beds. Other organizations, such as Ducks Unlimited, are interested in shallow lake restoration.



Dragonflies, such as this nymph of the extra-striped snaketail, are important indicator species for ecosystem health. *Photo by William Smith.*

8. Continue to develop a long-term inventory and monitoring program that includes the state's aquatic communities and the species dependent on aquatic systems. No agency can hope to sample all aquatic taxa in all state waters; an effective approach is to sample a statistically valid subset of waters using cost-effective biocriteria (e.g., indices of biotic integrity or abundance of environmentally sensitive species) and apply results to a well-developed waters classification system. Such a program will develop meaningful trend information and help identify problem areas in need of special protection or restoration efforts.

9. Consider the long-term, cumulative impacts of Department actions. Individual regulations or decisions may seem independent of one another, but in combination some may be inconsistent or have unintended impacts on the efforts of other programs or on aquatic resources. For example, any single macrophyte removal permit may seem minor, but the cumulative impact of many such permits may have significant effects on the ecosystem.

   a. Develop policy to establish an integrated, comprehensive approach to aquatic plant management. Three different programs are now involved in aquatic plant management, and their decisions are often based on different considerations. The Water Regulation program evaluates permits for structures such as sand blankets and mechanical weed control devices, the Lake Management program is responsible for permitting chemical and mechanical aquatic plant management proposals, and Fisheries Management is involved in the habitat issues related to both programs.

   b. Consider a pilot program for applying ecosystem management principles to selected aquatic regulatory and management programs. This would explore ways to correct problems arising when one program makes decisions that impact parts of the ecosystem managed by another program. This kind of approach might be easier to pilot for aquatic than terrestrial systems, because aquatic community boundaries are more easily defined, and they are typically already under Department management authority.

10. Work with the agricultural community and other public and private interests to address the effects of agriculture on aquatic ecosystems. Much is known about the impacts of erosion, nutrients, pesticides, and land use changes. Voluntary programs have not always been successful in mitigating these impacts but mandatory programs have not been popular. Incentives to alleviate the environmental effects of agricultural practices need greater attention in federal farm legislation and programs.

11. Emphasize critical aquatic habitat protection and restoration priorities in land acquisition and easement programs. Undeveloped shoreline areas deserve special consideration because these opportunities are rapidly declining.

12. Study the genetic composition of selected native fish species and modify fish stocking, transfer, and bait collecting policies if they appear detrimental to genetic diversity.

13. Take action at the state and federal levels to prevent the invasions of exotic species. Contingency plans would prepare the Department to be proactive when small infestations occur. Public education and awareness programs can help minimize the risk of importation and introduction.

14. Exercise extreme caution in implementing biological engineering to intensively manage the aquatic community. It is doubtful that such technology will be cost-effective or desirable when compared with the benefits of protecting naturally reproducing populations within self-sustaining ecosystems.

15. Support and conduct additional research to apply the principles of conservation biology to the management of aquatic communities and ecosystems. Many important questions remain unanswered, for instance: do rivers that have been fragmented by dams follow the principles of island biogeography? How does a lake's size affect its susceptibility to various kinds of disturbance? Is fish stocking of smaller waters more detrimental to biodiversity than stocking of larger waters?

Case Study

HABITAT RESTORATION FOLLOWING DAM REMOVAL ON THE MILWAUKEE RIVER AT WEST BEND

Contributed by Mike Staggs, John Lyons, and Kris Visser

There are over 50 dams in the Milwaukee River Basin, most holding back small impoundments of 50 acres or less. Because most of these impoundments originated as mill dams, they are located in the heart of urban areas and are valued by local residents for ice skating, waterfowl viewing, and their aesthetic qualities. Ecologically, however, these impoundments fragment fish habitat, create barriers to fish movement, may create thermal pollution problems, and typically have poor water quality as a result of sedimentation and related eutrophication. In some cases, the dams are more than a century old, creating safety concerns for their public and private owners. Thus, management of these dams and their associated impoundments poses an ecologically and socially complex problem in the Milwaukee River Basin.

In West Bend, a wooden dam was built across the river as early as 1870 to operate a woolen mill. In 1919 it was replaced by a concrete dam, which was operated privately for nearly 40 years to produce hydropower. The City took ownership of the dam in 1959. By 1987, the dam was in obvious need of removal or replacement. The City either had to remove the dam and restore the associated riverbed or replace the dam in conjunction with the construction of a new bridge.

The impoundment in the Milwaukee River at West Bend was drawn down in February 1988 to prepare for dam removal. *Photo by Paul Kanehl.*

A DNR team studied the 67-acre impoundment and found siltation; poor water quality; high turbidity; low recreational values due to shallow depth; a fish population heavily dominated by carp, suckers, and bluntnose minnows; and a lack of aquatic vegetation throughout the entire impoundment. Both upstream and downstream from the impoundment, where the river still flowed freely within its banks, the fish population was dominated by a variety of minnows, darters, crappie, bluegills and other panfish, and smallmouth and largemouth bass. In the river itself, carp, suckers, and bluntnose minnows were much less abundant than in the impoundment; carp made up 83% of the catch in the impoundment but only 23% above and below it.

After considerable public discussion, the City decided to accept the Department's recommendation to remove the dam, rehabilitate the riverbed, and stock gamefish. The goal was to restore self-sustaining habitat for smallmouth bass and other native fish species, to eliminate barriers to fish migration, to improve water quality, to create an urban park along the shoreline to provide recreational opportunity, and to use cost-effective methods to achieve these goals.

The dam was removed in May 1988. After dam removal, Department managers did some habitat improvement work throughout 1989 and 1990, including removing material from a portion of the channel area; placing logs, tree root masses, boulders, and similar materials underwater to create "instant" habitat; and rip-rapping some areas to prevent erosion. However, most of the formerly impounded area was allowed to recover naturally, without management.

The restoration produced 1.5 miles of free flowing river. Fish access to one mile of river upstream from the former impoundment was also regained. Floodplain areas of the former impoundment were developed as parkland, and oaks and maples were planted along the banks. Aquatic habitat quality improved dramatically. Aquatic vegetation quickly returned. Carp populations declined, and smallmouth bass and panfish populations increased. One threatened species, the greater redhorse, is now found in this restored area of the river.

This section of the Milwaukee River was restored following dam removal in May 1988. The formerly fragmented habitat is once again a free-flowing river with abundant aquatic vegetation and a diverse fish community. This photo was taken in June 1991. *Photo by Paul Kanehl.*

Literature Cited

Adler, K. 1968. Turtles from archeological sites in the Great Lakes region. Mich. Archaeol. 14(3-4):L147-163.

Allen, S.K., Jr., and R.J. Wattendorf. 1987. Triploid grass carp: status and management implications. Fisheries 12(4):20-24.

Armour, C.L., D.A. Duff, and W. Elmore. 1991. The effects of livestock grazing on riparian and stream ecosystems. Fisheries 16(1):6-11.

Arner, D., D. Robinette, J. Fresier, and M. Gray. 1979. Effects of channelization on the Luxapalila River on fish, aquatic invertebrates, water quality and fur bearers. Selected water resources abstracts, Vol. 12, W79-02636.

Bailey, R.G. 1989*a*. Ecoregions of the continents (map). U.S. Dep. Ag., For. Serv.

Bailey, R.G. 1989*b*. Explanatory supplement to the ecoregions map of the continents. Environ. Cons. 15(4):307-309.

Bailey, R.M. and G.R. Smith. 1981. Origin and geography of the fish fauna of the Laurentian Great Lakes basin. Can. J. Fish. and Aquat. Sci. 38(12):1539-61.

Bain, M.B., J.T. Finn, and H.E. Booke. 1988. Stream-flow regulation and fish community structure. Ecology 69:382-392.

Baker, F.C. 1928*a*. The fresh water Mollusca of Wisconsin: Part I. Gastropoda. Wis. Geol. Nat. Hist. Surv., Bulletin 70 Part I, Madison. 507 pp.

Baker, F.C. 1928*b*. The fresh water Mollusca of Wisconsin: Part II. Pelecypoda. Wis. Geol. Nat. Hist. Surv., Bulletin 70 Part II, Madison. 495 pp.

Baltz, D.M. and P.B. Moyle. 1993. Invasion resistance to introduced species by a native assemblage of California stream fishes. Ecol. Appl. 3:246-255.

Barton, D.R. and H.B.N. Hynes. 1978. Wave zone macrobenthos of the exposed Canadian shores of the St. Lawrence Great Lakes. J. Great Lakes Res. 4:27-45.

Barton, K. 1978. The other water pollution. Environment 20(5):12-20.

Becker, G.C. 1983. Fishes of Wisconsin. Univ. Wis. Press. 1052 pp.

Beckett, D.C. and M.C. Miller. 1982. Macroinvertebrate colonization of multiplate samplers in the Ohio River: the effect of dams. Can. J. Fish. Aquat. Sci. 39:1622-1627.

Benke, A.C., R.L. Henry, III, D.M. Gillespie, and R.J. Hunter. 1985. Importance of snag habitat for animal production in southeastern streams. Fisheries 10(5)8-13.

Benke, A.C., T.C. Van Arsdall, D.M. Gillespie, and F.K. Parrish. 1984. Invertebrate productivity in a subtropical blackwater river: the importance of habitat and life history. Ecol. Monogr. 54:25-63.

Bergquist, B.C. 1991. Extinction and natural recolonization of fish in acidified and limed lakes. Nordic J. Freshwater Res. 66:50-62.

Bergquist, S. 1986*a*. Point and nonpoint source control: Wisconsin waters need both. Wis. Nat. Resour. 10(1-suppl.):7-9.

Bergquist, S. 1986*b*. DNR's comprehensive watershed approach. Wis. Nat. Resour. 10(1-suppl.):20-29.

Berkman, H.E. and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. Env. Biol. Fish. 18(4):285-294.

Bernath, S.C., E.S. Verry, K.N. Brooks, and P.F. Ffolliott. 1982. Modeling water yield response to forest cover changes in northern Minnesota. pp. 385-399 *in* Canadian Hydrology Symposium, 14-15 June 1982, Frederickton, New Brunswick. Nat. Res. Council of Canada, Toronto.

Bernthal, T., G. Chism, F. Dreher, B. Emmons, and R. Schmidt. 1992. Looking forward: water resource issues on the Lac Courte Oreilles reservation - wild rice protection and enhancement. Institute for Environmental Studies, Water Resour. Manage., Univ. Wis. Madison.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon coast range. Water Resour. Res. 14:1011-1016.

Bilby, R.E. and J.W. Ward. 1989. Changes in characteristics and function of woody debris with increasing size of streams in western Washington. Trans. Am. Fish. Soc. 118:368-78.

Billington, N. 1990. Mitochondrial variation in Wisconsin walleye populations. Report to Wis. Dep. Nat. Resour. 4 Sep 1990.

Billington, N. and P.D.N. Hebert. 1988. Mitochondrial DNA variation in Great Lakes walleye (*Stizostedion vitreum*) populations. Can. J. Fish. and Aquat. Sci. 45:643-54.

Billington, N., P.D.N. Hebert, and R.D. Ward. 1988. Evidence of introgressive hybridization in the genus *Stizostedion*: interspecific transfer of mitochondrial DNA between sauger and walleye. Can. J. Fish. Aquat. Sci. 45:2035-2041.

Birge, W.J., A. Black, and R.A. Kuehne. 1980. Effects of organic compounds on amphibian reproduction. Univ. Kentucky, Lexington, Water Resour. Res. Inst., Res. Rep. No.121.

Birge, W.J., A. Black, A.G. Westerman, and J.E. Hudson. 1979. The effects of mercury on reproduction of fish and amphibians. pp. 629-55 *in* J.R. Nriagu, ed. The biogeochemistry of mercury in the environment. Elsevier/North Holland Biomedical Press, Amsterdam.

Blumberg, A.F. and D.M. Di Toro. 1990. Effects of climate warming on dissolved oxygen concentrations in Lake Erie. Trans. Am. Fish. Soc. 119(2):210-23.

Bode, J., S. Borman, S. Engel, D. Helsel, F. Koshere, and S. Nichols. 1993. Eurasian water milfoil in Wisconsin: a report to the legislature. Wis. Dep. Nat. Resour., Madison.

Bovee, K.D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. Instream flow information paper No. 12. U.S. Fish and Wildl. Serv. FWS/OBS-82/26. Ft. Collins, Col.

Breitburg, D.L. 1988. Effects of turbidity on prey consumption by striped bass larvae. Trans. Am. Fish. Soc. 117:72-77.

Brooks, J.L. and S.I. Dodson. 1965. Predation, body size, and the composition of plankton. Science 150:28-35.

Bryan, M.D. and D.L. Scarnecchia. 1992. Species richness, composition, and abundance of fish larvae and juveniles inhabiting natural and developed shorelines of a glacial Iowa lake. Env. Biol. Fish. 35:329-341.

Bundy, W.F. 1882. A list of the Crustacea of Wisconsin, with notes on some new or little known species. Trans. Wis. Acad. Sci. Arts Lett. 5:177-184.

Burton, T.M. and G.E. Likens. 1975. Salamander populations and biomass in the Hubbard Brook Experimental Forest, New Hampshire. Copeia 1975(3):541-46.

Camargo, J.A. 1992. Macroinvertebrate responses along the recovery gradient of a regulated river (Spain) receiving an industrial effluent. Arch. Environ. Contam. Toxicol. 23:324-332.

Carlander, H.B. 1954. History of fish and fishing in the Upper Mississippi River. Upper Miss. River Conserv. Comm. 96 pp.

Carline, R.F. and O.M. Brynildson. 1977. Effects of hydraulic dredging on the ecology of native trout populations in Wisconsin spring ponds. Wis. Dep. Nat. Resour. Tech. Bull. No. 98.

Carlson, H. and L.M. Andrews. 1977. Surface water resources of Marathon County. Wis. Dep. Nat. Resour., Madison. 81 pp.

Chadwick, J.W., S.P. Canton, and R.L. Dent. 1986. Recovery of benthic invertebrate communities in Silver Bow Creek, Montana, following improved metal mine waste-water treatment. Water, Air, and Soil Pollution 28:427-438.

Clark, C.W. 1991. Economic biases against sustainable development. *in* R. Costanza, ed. Ecological economics: the science and management of sustainability. 320 pp.

Clark, R.D., Jr., and B. Huang. 1985. Conflict between sportfishing, commercial fishing, and rehabilitation of lake trout in Lake Michigan. N. Am. J. Fish. Manage. 5:261-76.

Coble, D.W. 1982. Fish populations in relation to dissolved oxygen in the Wisconsin River. Trans. Am. Fish. Soc. 111:612-623.

Coble, D.W. 1988. Effects of angling on bluegill populations: management implications. N. Am. J. Fish. Manage. 8(3):277-283.

Colby, P.J., P.A. Ryan, D.H. Schupp, and S.L. Serns. 1987. Interactions in north-temperate lake fish communities. Can. J. Fish. and Aquat. Sci. 44(Supp. 2):104-28.

Cook, D.G. and M.G. Johnson. 1974. Benthic macroinvertebrates of the St. Lawrence Great Lakes. J. Fish. Res. Bd. Can. 31:763-782.

Courtney, W.R., Jr., and P.B. Moyle. 1992. Crimes against biodiversity: the lasting legacy of fish introductions. Trans. 57th N. A. Wildl. and Nat. Res. Conf. 1992:365-372.

Creaser, E.P. 1932. The decapod crustaceans of Wisconsin. Trans. Wis. Acad. Sci. Arts Lett. 27:321-338.

Cuffney, T.F., J.B. Wallace, and J.R. Webster. 1984. Pesticide manipulation of a headwater stream: invertebrate responses and their significance for ecosystem processes. Freshw. Invert. Biol. 3:153-171.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of stream benthos to fine sediment deposition versus transport. Can. J. Zool. 64:1345-1351.

Cummins, K.W., G.W. Minshall, J.R. Sedell, C.E. Cushing, and R.C. Petersen. 1984. Stream ecosystem theory. Verh. Int. Ver. Limnol. 22:1818-1827.

Cushman, R.M. 1985. Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. N. Am. J. Fish. Manage. 5:330-339.

Dermott, R. 1978. Benthic diversity and substrate-fauna associations in Lake Superior. J. Great Lakes Res. 4:505-512.

Detenbeck, N.E., P.W. Devore, G.J. Niemi, and A. Lima. 1992. Recovery of temperate stream fish communities from disturbance: a review of case studies and synthesis of theory. Environ. Manage. 16:33-53.

Dillon, P.J. and F.H. Rigler. 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. Can. J. Fish. and Aquat. Sci. 32:1519-1531.

Doeg, T. J., R. Marchant, M. Douglas, and P.S. Lake. 1989. Experimental colonization of sand, gravel and stones by macroinvertebrates in the Acheron River, southeastern Australia. Freshw. Biol. 22:57-64.

Dorff, C.J. 1990. The effects of wetland drawdown on Blanding's turtle behavior and ecology. Univ. Minn., St. Paul. [unpubl. ms.]

Downs, W. 1984. Fish of Lake Superior. Univ. Wis. Sea Grant Inst. WIS-SG-76-124, Sep 1984.

Downs, W. 1986. Fish of Lake Michigan. Univ. Wis. Sea Grant Inst. WIS-SG-74-121, Sep 1986.

Dudley, T. and N.H. Anderson. 1982. A survey of invertebrates associated with wood debris in aquatic habitats. Melanderia 39:1-21.

Duncan, R.E. and P.A. Thiel. 1983. A survey of the mussel densities in Pool 10 of the Upper Mississippi River. Wis. Dep. Nat. Resour. Tech. Bull. No. 139, Madison. 14 pp.

Dundee et al. 1992. Report to the Louisiana reptile and amphibian task force. La. Dep. Wildl. and Fish., Hab. Conserv. Div. [unpubl. data.]

Eck, G.W. and E.H. Brown. 1985. Lake Michigan's capacity to support lake trout (*salvelinus namaycush*) and other salmonids: an estimate based on the status of prey populations in the 1970s. Can. J. Fish. and Aquat. Sci. 42:449-54.

Eckblad, J.W. 1986. The ecology of pools 11-13 of the Upper Mississippi River: a community profile. U.S. Fish. and Wildl. Serv., Biol. Rep. 85 (7.8). 90 pp.

Ecological Analysts. 1981. Survey of freshwater mussels (Pelecypoda: Unionacea) at selected sites in Pools 11 through 24 of the Mississippi River. U.S. Army Corps of Engineers, Rock Island District, Rock Island, Ill. 188 pp.

Eddy, S. and J.C. Underhill. 1974. Northern fishes. Univ. Minn. Press, Minneapolis. 414 pp.

Ellis, M.M. 1936. Erosion silt as a factor in aquatic environments. Ecology 17(1):29-42.

Engel, S. and S.A. Nichols. 1994. Restoring Rice Lake at Milltown, Wisconsin. Wis. Dep. Nat. Resour. Tech. Bull. No. 186. 42 pp.

Extence, C.A. 1981. The effect of drought on benthic invertebrate communities in a lowland river. Hydrobiologia 83:217-224.

Fago, D. 1992. Distribution and relative abundance of fishes in Wisconsin. VIII. Final Report. Wis. Dep. Nat. Resour. Tech. Bull. No. 175.

Fago, D. and A.B. Hauber. 1993. Black redhorse, *Moxostoma duquesnei* rediscovered in Wisconsin. Can. Field-Naturalist 107(3):351-352.

Fannucchi, G.T., W.A. Fannucchi, and S. Craven. 1986. Wild rice in Wisconsin: its ecology and cultivation. Univ. Wis. Extension Pub. No. G3372. 8 pp.

Fausch, K.D., J. Lyons, J.R. Karr, and P.L. Angermeier. 1990. Fish communities as indicators of environmental degradation. Am. Fish. Soc. Symp. 8:123-44.

Fisher, S.G. and A. LaVoy. 1972. Differences in littoral fauna due to fluctuating water levels below a hydroelectric dam. J. Fish. Res. Bd. Can. 29:1472-1476.

Forbes, A.M. 1985. Summary of survey data for smallmouth bass in Wisconsin streams, 1952-80. Wis. Dep. Nat. Resour. Bur. Res. Rep. No. 133.

Forney, J.L. 1974. Interactions between yellow perch abundance, walleye predation and survival of alternate prey in Oneida Lake, New York. Trans. Am. Fish. Soc. 103(1):15-24.

Fredricksen, R.L. 1970. Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds. U.S. Dep. Ag., For. Ser. Res. Pap. PNW-104.

Fremling, C.R., J.L. Rasmussen, R.E. Sparks, S.P. Cobb, C.F. Bryan, and T.O. Claflin. 1989. Mississippi River fisheries: a case history. pp. 309-351 *in* D.P. Dodge, ed. Proceedings of the international large river symposium. Can. Spec. Pub. Fish. Aquat. Sci. No. 106. 628 pp.

Frey, D.G. 1963. Wisconsin: the Birge-Juday era. *in* D.G. Frey, ed. Limnology of North America. Univ. Wis. Press.

Fuller, S.L.H. 1974. Clams and mussels (Mollusca:Bivalvia) *in* C.W. Hart, Jr., and S.L.H. Fuller, eds. Pollution ecology of freshwater invertebrates. Academic Press, Inc., New York. 389 pp.

Gardner, M.B. 1981. Effects of turbidity on feeding rates and selectivity of bluegills. Trans. Am. Fish. Soc. 110:446-450.

Garton, D.W. and D.J. Berg. 1990. Occurrence of *Bythotrephes cederstroemi* (Schoedler 1877) in Lake Superior, with evidence of demographic variation within the Great Lakes. J. Great Lakes Res. 16:148-152.

Gersich, F. M. and M. A. Brusven. 1981. Insect colonization rates in near-shore regions subjected to hydroelectric power peaking flows. J. Freshw. Ecol. 1:231-236.

Giesy, J.P., J.P. Ludwig, and D.E. Tillit. 1994. Deformities in birds of the Great Lakes region: assigning causality. Environ. Sci. Technol. 28(3):128-135.

Gray, J.R.A. and J.M. Edington. 1969. Effect of woodland clearance on stream temperature. Can. J. Fish. Aquat. Sci. 26:399-403.

Greene, C.W. 1935. The distribution of Wisconsin fishes. State of Wis. Conserv. Comm., Madison. 235 pp.

Gresswell, R.E. and J.D. Varley. 1988. Effects of a century of human influence on the cutthroat of Yellowstone Lake. Am. Fish. Soc. Symp. 4:45-52.

Hall, R.J. and E. Kolbe. 1980. Bioconcentrations of organophosphorus pesticides to hazardous levels by amphibians. J. Toxicol. Environ. Health 6:853-60.

Handford, P., G. Bell, and T. Reimchen. 1977. A gillnet fishery considered as an experiment in artificial selection. Can. J. Fish. and Aquat. Sci. 34:954-61.

Hansen, M.J. 1989. A walleye population model for setting harvest quotas. Wis. Dep. Nat. Resour. Bur. Fish. Manage. Rep. No. 143. 12 pp.

Hansen, M.J., M.D. Staggs, and M.H. Hoff. 1991. Derivation of safety factors for setting harvest quotas on adult walleyes from past estimates of abundance. Trans. Am. Fish. Soc. 120:620-628.

Hansmann, E.W. and H.K. Phinney. 1973. Effects of logging on periphyton in coastal streams of Oregon. Ecology 54:194-198.

Hauber, A. 1989. Impact of continuous fishing on the walleye population of Lake DuBay, Marathon County, Wisconsin, 1983-84. Wis. Dep. Nat. Resour. Bur. Fish. Manage. Rep. No. 142.

Hazelwood, E. 1970. Frog pond contaminated. Brit. J. Herpe. 4:177-85.

Healey, M.C. 1980. Growth and recruitment in experimentally exploited lake whitefish (*Coregonus clupeaformis*) populations. Can. J. Fish. and Aquat. Sci. 37:255-67.

Heath, D.J. 1992. A preliminary survey for rare reptiles at selected Wisconsin Valley Improvement Company impoundment projects located in Wisconsin. Wis. Dep. Nat. Resour., North Central District Headquarters, Rhinelander.

Heath, D.J. 1993*a*. Justification and feasibility analysis for freshwater mussel reestablishment upstream of the Prairie du Sac dam (FERC project #11162), Wisconsin River, Wisconsin. Wis. Dep. Nat. Resour., North Central District Headquarters, Rhinelander.

Heath, D.J. 1993*b*. A preliminary survey for rare reptiles at the Wausau hydroelectric project (FERC # 1999) located in Marathon county, Wisconsin. Wis. Dep. Nat. Resour., North Central District Headquarters, Rhinelander.

Heath, D.J. 1993*c*. A preliminary survey for the Wisconsin state threatened Blanding's turtle (*Emydoidea blandingii*) at the Kings (FERC #2239) (Lake Alice) hydroelectric project located near Tomahawk, Wisconsin. Wis. Dep. Nat. Resour., North Central District Headquarters, Rhinelander.

Hebert, P.N.D., C.C. Wilson, M.H. Murdoch, and R. Lazar. 1991. Demography and ecological impacts of the invading mollusc *Dreissena polymorpha*. Can. J. Zool. 69:405-409.

Helms, D.R. 1974. Shovelnose sturgeon in the Mississippi River, Iowa. Iowa Conserv. Comm. Comm. Fish. Res. Tech. Ser. 74-3. 68 pp.

Hewlett, J.D. and J.C. Fortson. 1982. Stream temperature under an inadequate buffer strip in the southeast Piedmont. Water Resour. Bull. 18:983-988.

Hibbert, A.R. 1969. Water yield changes after converting a forested catchment to grass. Water Resour. Res. 5:634-640.

Hicks, B.J., R.L. Beschta, and R.D. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resour. Bull. 27:217-226.

Hill, D.K. and J.J. Magnuson. 1990. Potential effects of global warming on the growth and prey consumption of Great Lakes fish. Trans. Am. Fish. Soc. 119:(2):265-75.

Hilsenhoff, W.L. 1971. Changes in the downstream insect and amphipod fauna caused by an impoundment with a hypolimnion drain. Ann. Am. Entomol. Soc. 64:743-746.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomol. 20:31-39.

Hindar, K., R. Ryman, and F. Utter. 1991. Genetic effects of cultured fish on natural fish populations. Can. J. Fish. and Aquat. Sci. 48:945-57.

Hobbs, H.H., III and J.P. Jass. 1988. The crayfishes and shrimp of Wisconsin. Milwaukee Public Museum, Milwaukee, Wis. 177 pp.

Holland, L.E. 1987. Effect of brief navigation-related dewaterings on fish eggs and larvae. N. Am. J. Fish. Manage. 7:145-147.

Holland, L.E. and M.L. Huston. 1984. Relationship of young-of-the-year northern pike to aquatic vegetation types in backwaters of the Upper Mississippi River. N. Am. J. Fish. Manage. 4:514-522.

Hornbeck, J.W., R.S. Pierce, and C.A. Federer. 1970. Streamflow changes after forest clearing in New England. Water Resour. Res. 6:1124-1132.

Jackivicz, T.P., Jr., and L.N. Kuzminski. 1973. A review of outboard motor effects on the aquatic environment. J. Water Poll. Con. Fed. 45(8):1759-1770.

Johnson, C.E. 1971. Factors affecting fish spawning. Wis. Cons. Bull. 36(4):16-17.

Johnson, F.H. 1961. Walleye egg survival during incubation on several types of bottom in Lake Winnibigoshish, Minnesota and connecting waters. Tran. Am. Fish. Soc. 90:312-322.

Johnson, T.B. and D.O. Evans. 1990. Size-dependent winter mortality of young-of-the-year white perch: climate warming and invasion of the Laurentian Great Lakes. Trans. Am. Fish. Soc. 119(2):301-13.

Jude, J.D. and F.J. Tesar. 1985. Recent changes in the inshore forage fish of Lake Michigan. Can. J. Fish. Aquat. Sci. 42:1154-57.

Junk, W.J., P.B. Bayley, and R.E. Sparks. 1989. The flood pulse concept in river-floodplain systems. pp. 110-127 *in* D.P. Dodge, ed. Proceedings of the international large river symposium. Can. Spec. Publ. Fish. Aquat. Sci. No. 106.

Kallemeyn, L.W. 1987. Correlations of regulated lake levels and climatic factors with abundance of young-of-the-year walleye and yellow perch in four lakes in Voyageurs National Park. N. Am. J. Fish. Manage. 7:513-521.

Kanehl, P. and J. Lyons. 1992. Impacts of in-stream sand and gravel mining on stream habitat and fish communities, including a survey on the Big Rib River, Marathon county, Wisconsin. Wis. Dep. Nat. Resour. Bur. Res. Rep. No. 155. 32 pp.

Kapuscinski, A.R. and L.D. Jacobson. 1987. Genetic guidelines for fisheries management. Univ. Minn. Sea Grant Res. Rep. No. 17.

Kapuscinski, A.R.D. and J.E. Lannan. 1986. A conceptual genetic fitness model for fisheries management. Can. J. Fish. Aquat. Sci. 43:1606-16.

Kapuscinski, A.R. and D.P. Philipp. 1988. Fisheries genetics: issues and priorities for research and policy development. Fisheries 13:4-10.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21-27.

Karr, J.R., L.A. Toth, and D.R. Dudley. 1985. Fish communities of midwestern rivers: a history of degradation. BioScience 35(2):90-95.

Keddy, P.A. 1983. Shoreline vegetation in Axe Lake, Ontario: effects of exposure on zonation patterns. Ecology 64:331-344.

Keddy, P.A. and A.A. Reznicek. 1986. Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. J. Great Lakes Res. 12:25-36.

Kempinger, J.J., W.S. Churchill, G.R. Priegel, and L.M. Christensen. 1975. Estimates of abundance, harvests, and exploitation of the fish population of Escanaba Lake, Wisconsin, 1946-69. Wis. Dep. Nat. Resour. Tech. Bull. No. 84. 30 pp.

Kitchell, J.F. 1992. Introduction: the rationale and goals for food web management in Lake Mendota. *in* J.F. Kitchell, ed. Food web management: a case study of Lake Mendota. Springer-Verlag, New York. 553 pp.

Kitchell, J.F. and P.R. Sanford. 1992. Paleolimnological evidence of food web dynamics in Lake Mendota. *in* J.F. Kitchell, ed. Food web management: a case study of Lake Mendota. Springer-Verlag, New York. 553 pp.

Kleinert, S.J., P.E. Degurse, and J. Ruhland. 1974. Concentration of metals in fish. Wis. Dep. Nat. Resour. Tech. Bull. No. 74.

Konrad, J.G. and S.J. Kleinert. 1974. Removal of metals from waste waters by municipal sewage treatment plants. Wis. Dep. Nat. Resour. Tech. Bull. No. 74.

Korth, R. 1993. About jewels, magic and lakes. Lake Tides 18(2):4-5.

Kroger, R.L. 1973. Biological effects of fluctuating water levels in the Snake River, Grand Teton National Park, Wyoming. Am. Midl. Nat. 89:478-481.

Kuhm, H.W. 1928. Wisconsin Indian fishing - primitive and modern. Wis. Archeol. 7:61-115.

Lagler, K.F., A.S. Hazzard, W.E. Hazen, and W.A. Tompkins. 1950. Outboard motors in relation to fish behavior, fish production, and angling success. Proc. 15th N. Am. Wild. Conf., March 6-9, 1950: 280-303.

Larson, G.L. and S.E. Moore. 1985. Encroachment of exotic rainbow trout into stream populations of native brook trout in the southern Appalachian mountains. Trans. Am. Fish. Soc. 114:195-203.

Lathrop, R.C., P.W. Rasmussen, and D.R. Knauer. 1990. Walleye mercury concentrations in Wisconsin lakes. Pap. presented at International Conference on Mercury, Gavle, Sweden, 11 June 1990.

Lawrie, A.H. and J.F. Rahrer. 1973. Lake Superior - a case history of the lake and its fisheries. Great Lakes Fish. Comm. Tech. Rep. No. 19.

Lee, D.S., C.R. Gilbert, C.H. Hocutt, R.E. Jenkins, D.E. McAllister, and J.R. Stauffer, Jr. 1980. Atlas of North American Fishes. North Carolina State Museum of Nat. Hist., Raleigh.

Lemly, A.D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. Hydrobiologia 87:229-245.

Leonard, P.M. and D.J. Orth. 1988. Use of habitat guilds of fishes to determine instream flow requirements. N. Am. J. Fish. Manage. 8(4):399-409.

Liddle, M.J. and H.R.A. Scorgie. 1980. The effects of recreation on freshwater plants and animals: a review. Biol. Cons. 17:183-206.

Lillie, R.A. and J.W. Mason. 1983. Limnological characteristics of Wisconsin lakes. Wis. Dep. Nat. Resour. Tech. Bull. No. 138. 116 pp.

Linder, G., J. Barbitta, and T. Kwaiser. 1990. Short-term amphibian toxicity tests and paraquat toxicity assessment. pp. 189-98 *in* W.G. Landis and W.H. van der Schalie, eds. Aquatics toxicology and risk assessment. Vol. 13. Am. Soc. for Testing and Materials, Philadelphia.

Lloyd, D.S., J.P. Koenings, and J.D. LaPerriere. 1987. Effects of turbidity in fresh waters of Alaska. N. Am. J. Fish. Manage. 7:18-33.

Lubinski, K.S. and S. Gutreuter. 1993. Ecological information and habitat rehabilitation on the Upper Mississippi River. pp. 87-100 *in* L.W. Hesse, C.B. Stalnaker, N.G. Benson, and J.R. Zuboy, eds. Proceedings of the symposium on restoration planning for the rivers of the Mississippi River ecosystem. U.S. Dep. Int., Nat. Biol. Surv., Biol. Rep. No. 19. 502 pp.

Ludwig, D., R. Hilborn, and C. Walters. 1993. Uncertainty, resource exploitation, and conservation: lessons from history. Science 260:17.

Lynch, J.A., G.B. Reshez, and E.S. Corbett. 1984. Thermal alteration of streams draining clear-cut watersheds: quantification and biological implications. Hydrobiologia 111:161-169.

Lyons, J. 1989*a*. Correspondence between the distribution of fish assemblages in Wisconsin streams and Omernik's ecoregions. Am. Midl. Nat. 122:163-82.

Lyons, J. 1989*b*. Changes in the abundance of small littoral-zone fishes in Lake Mendota, Wisconsin. Can. J. Zool. 67:2910-2916.

Lyons, J. 1992. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. U.S. Dep. Ag., For. Serv., N. Cent. For. Exp. Stn., Gen. Tech. Rep. NC-149. 51 pp.

Lyons, J. 1993. Status and biology of paddlefish (*Polyodon spathula*) in the lower Wisconsin river. Trans. Wis. Acad. Sci. Arts Let. 81:123-135.

Lyons, J., A.M. Forbes, and M.D. Staggs. 1988. Fish species assemblages in southwestern Wisconsin streams with implications for smallmouth bass management. Wis. Dep. Nat. Resour. Tech. Bull. No. 161.

Mack, W.N. and F.M. D'Itri. 1973. Pollution of a marina area by watercraft use. J. Water Poll. Cont. Fed. 45(1):97-104.

Mackie, G.L. 1991. Biology of the exotic zebra mussel, *Dreissena polymorpha*, in relation to native bivalves and its potential impact in Lake St. Clair. Hydrobiologia 219:251-268.

Magnuson, J.J. 1976. Managing with exotics—a game of chance. Trans. Am. Fish. Soc. 105(1):1-9.

Magnuson, J.J., J.D. Meisner, and D.K. Hill. 1990. Potential changes in thermal habitat of Great Lakes fish after global climate warming. Trans. Am. Fish. Soc. 119(2):254-64.

Mandrak, N.E. 1989. Probable invasions of the Great Lakes by new species due to climate warming. J. Great Lakes Res. 15:306-16.

Manning, R.E. 1979. Impacts of recreation on riparian soils and vegetation. Wat. Res. Bull. 15(1):30-43.

Marron, M. 1994. Mississippi River commercial fishing statistics. Wis. Dep. Nat. Resour. Bur. Fish. Manage. Stat. Rep. 26 pp.

Marshall, T.R., R.A. Ryder, C.J. Edwards, and G.R. Spangler. 1987. Using the lake trout as an indicator of ecosystem health - application of the dichotomous key. Great Lakes Fish. Comm. Tech. Rep. No. 49.

Martinez, P.J., T.E. Chart, M.A. Trammell, J.G. Wullschleger, and E.P. Bergersen. 1994. Fish species composition before and after construction of a main stem reservoir on the White River, Colorado. Environ. Biol. Fish. 40:227-239.

Mason, J.W., D.J. Graczyk, and R.A. Kerr. 1991. Effects of runoff on smallmouth bass populations in four southwestern Wisconsin streams. pp. 28-38 *in* D.C. Jackson, ed. The First International Smallmouth Bass Symposium. Mississippi Agricultural and Forestry Experiment Station, Miss. State Univ.

Mathiak, H.A. 1979. A river survey of the Unionid mussels of Wisconsin, 1973-1977. Sand Shell Press, Horicon, Wis. 75 pp.

McCarraher, D.B. and R.E. Thomas. 1972. Ecological significance of vegetation to northern pike, *Esox lucius*, spawning. Trans. Am. Fish. Soc. 101(3):560-563.

McCormick, M.J. 1990. Potential changes in thermal structure and cycle of Lake Michigan due to global warming. Trans. Am. Fish. Soc. 119(2):183-94.

McInerny, M.C., R.L. Glazer, G.W. Zarling, J.E. Fulton, J.S. Otis, and K.S. Guise. 1991. Genetic description of walleye stocks in Minnesota. Minn. Dep. Nat. Resour. Invest. Rep. No. 411.

McMahon, R.F. 1991. Mollusca: Bivalvia. pp. 315-400 *in* J.H. Thorp and A.P. Covich, eds. Ecology and classification of North American freshwater invertebrates, Academic Press, San Diego, California. 911 pp.

Meffe, G.K. 1987. Conserving fish genomes: philosophies and practices. Environ. Biol. Fish. 18:3-9.

Meffe, G.K. 1991. Failed invasion of southeastern blackwater stream by bluegills: implications for conservation of native communities. Trans. Am. Fish. Soc. 120:333-38.

Meisner, J.D. 1990. Potential loss of thermal habitat for brook trout, due to climatic warming, in two southern Ontario streams. Trans. Am. Fish. Soc. 119(2):282-91.

Merendino, M.T., L.M. Smith, H.R. Murkin, and R.L. Pererson. 1990. The response of prairie wetland vegetation to seasonality of drawdowns. Wildl. Soc. Bull. 18:245-251.

Meyer, M.A. 1994. Mercury in wildlife: Wisconsin acid deposition monitoring and evaluation program, 1993 annual report. Wis. Dep. Nat. Resour., Publ. AM-132-94, p. 87-103. Madison.

Milhous, R.T., D.L. Wegner, and T. Waddle. 1981. User's guide to the physical habitat simulation system. Instream Flow Information Paper No. 11. U.S. Fish and Wildl. Serv., FWS/OBS-81/43. Ft. Collins, Col.

Miller, R.R., J.D. Williams, and J.E. Williams. 1989. Extinctions of North American fishes during the past century. Fisheries 14:22-38.

Minckley, W.L. and J.N. Rinne. 1985. Large woody debris in hot desert streams: an historical review. Desert Plants 7(3)142-53.

Minshall, G.W. et al. 1985. Developments in stream ecosystem theory. Can. J. Fish. Aquat. Sci. 42:1045-1055.

Minton, S.R., Jr. 1972. Amphibians and reptiles of Indiana. Indiana Acad. Sci. Monogr. 3. 346 pp.

Morizot, D.C., S.W. Calhoun, L.L. Clepper, M.E. Schmidt, J.H. Williamson, and G.J. Carmichael. 1991. Multispecies hybridization among native and introduced centrarchid basses in central Texas. Trans. Am. Fish. Soc. 120:283-289.

Morton, J.W. 1977. Ecological effect of dredging and dredge spoil disposal: a literature review. U.S. Fish and Wild. Serv., Tech. Pap. 94, 33 pp.

Mosindy, T.A., W.T. Momot, and P.J. Colby. 1987. Impact of angling on the production and yield of mature walleyes and northern pike in a small boreal lake in Ontario. N. Am. J. Fish. Manage. 7:493-501.

Moss, B. 1977. Conservation problems in the Norfolk broads and rivers of East Anglia, England - phytoplankton, boats and the causes of turbidity. Biol. Cons. 12:95-114.

Mossman, M.J. and R. Hine. 1984. The Wisconsin frog and toad survey: establishing a long-term monitoring program. Wis. Dep. Nat. Resour. Bur. Endangered Resour. Rep. No. 9. 24 pp.

Mossman, M.J. and R. Hine. 1985. Wisconsin's frog and toad survey. Wis. Dep. Nat. Resour. Bur. Endangered Resour. Rep. No. 16. 16 pp.

Mossman, M.J. and J. Huff. 1990. Results of Wisconsin's frog and toad survey 1984-1989. [unpubl. rep.]

Moyle, P.B. 1991. Ballast water introductions. Fisheries 16(1):4-6.

Murphy, M.L. and K.V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. N. Am. J. Fish. Manage. 9:427-36.

Nalepa, T.F. 1989. Estimates of macroinvertebrate biomass in Lake Michigan. J. Great Lakes Res. 15:437-443.

Napoli, J. 1975. The coasts of Wisconsin. Univ. Wis. Sea Grant WIS-SG-75-122, March 1975.

Narf, R.P. 1985. Aquatic insect colonization and substrate changes in a relocated stream segment. Great Lakes Entomol. 18(2):83-92.

Neel, J.K. 1963. Impact of reservoirs. pp. 575-593 *in* D.G. Frey, ed. Limnology in North America. Univ. Wis. Press, Madison.

Neves, R.J. and J.C. Widlak. 1987. Habitat ecology of juvenile freshwater mussels (Bivalvia:Unionidae) in a headwater stream in Virginia. Amer. Malacol. Bull. 5:1-7.

Newbold, J.D., D.C. Erman, and K.B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. Can. J. Fish. Aquat. Sci.37:1076-1085.

Newcombe, C.P. and D.D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. N. Am. J. Fish. Manage. 11:72-82.

Ney, J.J. and M. Mauney. 1981. Impact of a small impoundment on benthic macroinvertebrate and fish communities of a headwater stream in the Virginia Piedmont. Am. Fish. Soc. Warmwater Streams Symposium, pp. 102-112.

Niemi, G.J. et al. 1990. Overview of case studies on recovery of aquatic systems from disturbance. Environ. Manage. 14(5):571-587.

Nilles, E. 1993. Emeralds, pygmies, skimmers, and snaketails: rare dragonflies in Wisconsin. The Niche 6(1):1-4.

Noel, D.S., C.W. Martin, and C.A. Federer. 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. Environ. Manage. 10:661-670.

Nuhfer, A.J. and G.R. Alexander. 1991. Growth, survival and vulnerability of angling of three wild brook trout strains exposed to different levels of angler exploitation over time. Mich. Dep. Nat. Resour. Fish. Res. Rep. No. 1973.

Olsen, T.M., D.M. Lodge, G.M. Capelli, and R.J. Houlihan. 1991. Mechanisms of impact of an introduced crayfish (*Orconectes rusticus*) on littoral cogeners, snails and macrophytes. Can. J. Fish. Aquat. Sci. 48:1853-1861.

Omernik, J.M. and A.L. Gallant. 1988. Ecoregions of the upper midwest states. EPA/600/3-88/037.

Pajak, P. 1991. Fisheries implications of the 1990 Farm Bill. Fisheries 16(4):4.

Panek, F.M. 1979. Cumulative effects of small modifications to habitat. Fisheries 4(2):54-57.

Pearce, A.J. and L.K. Rowe. 1979. Forest management effects on interception, evaporation and water yield. J. Hydrol. 18:73-87.

Pechmann, J.H.K., D.E. Scott, R.D. Semlitsch, J.P. Caldwell, L.J. Vitt, and J.W. Gibbons. 1991. Declining amphibian populations: the problem of separating human impacts from natural fluctuations. Science 253:892-95.

Penaloza, L.J. 1989. Wisconsin recreation survey - 1986. Wis. Dep. Nat. Resour. Tech. Bull. No. 167. 24 pp.

Penaloza, L.J. 1991. Boating pressure on Wisconsin's lake and rivers. Wis. Dep. Nat. Resour. Tech. Bull. No. 174. 52 pp.

Peterman, R.M. 1990. Statistical power analysis can improve fisheries research and management. Can. J. Fish. Aquat. Sci. 47:2-15.

Peterman, R.M. and M.J. Bradford. 1987. Statistical power of trends in fish abundance. Can. J. Fish. Aquat. Sci. 44:1879-1889.

Pflieger, W.L. 1971. A distributional study of Missouri River fishes. Univ. Kans., Lawrence. Publ. Mus. Nat. Hist. 20(3):225-570.

Philipp, D.P. 1991. Genetic implications of introducing Florida largemouth bass. Can. J. of Fish. and Aquat. Sci. 48(Suppl. 1):58-65.

Philipp, D.P., W.F. Childers, and G.S. Whitt. 1983. A biochemical genetic evaluation of the northern and Florida subspecies of largemouth bass. Trans. Am. Fish. Soc. 112:1-20.

Philipp, D.P., J.M. Epifanio, and M.J. Jennings. 1991. AFS position statement - Protecting native fish stocks: the elimination of stock transfers (draft). Ill. Nat. Hist. Surv., Cent. for Aquat. Ecol. [memo 26 Aug 1991]

Phillip, D.P. and G.S. Whitt. 1991. Survival and growth of northern, Florida and reciprocal F1 hybrid largemouth bass in central Illinois. Trans. Am. Fish. Soc. 120:58-64.

Policansky, D. 1993. Fishing as a cause of evolution in fishes. pp. 2-18 *in* T.K. Stokes, J.M. McGlade and R. Law, eds. The exploitation of evolving resources. Lecture Notes in Bio-Mathematics Volume 99. Springer-Verlag, Berlin.

Radford, D.S. and R. Hartland-Rowe. 1971. A preliminary investigation of bottom fauna and invertebrate drift in an unregulated and a regulated stream in Alberta. J. Appl. Ecol. 8:883-903.

Regier, H.A., J.J. Magnuson, and C.C. Coutant. 1990. Introduction to proceedings: symposium on effects of climate change on fish. Trans. Am. Fish. Soc. 119 (2)173-75.

Reid, D.M. and W.T. Momot. 1985. Evaluation of pulse fishing for the walleye, *Stizostedion vitreum vitreum*, in Henderson Lake, Ontario. J. Fish Biol. 27(A):235-51.

Ritchie, J.C. 1972. Sediment, fish and fish habitat. J. Soil and Water Cons. 27:124-125.

Robel, R.J. 1961. Water depth and turbidity in relation to growth of Sago Pondweed. J. Wild. Manage. 25(4):436-438.

Robins, C.R., R.M. Bailey, C.E. Bond, J.R. Brooker, E.A. Lachner, R.N. Lea, and W.B. Scott. 1991. Common and scientific names of fishes from the United States and Canada. Am. Fish. Soc. Spec. Publ. No. 20. 183 pp.

Rosenberg, D.M. and A.P. Wiens. 1978. Effects of sediment addition on macrobenthic invertebrates in a northern Canadian river. Water Res. 12:753-763.

Ryder, R.A. and C.J. Edwards. 1985. A conceptual approach for the application of biological indicators of ecosystem quality in the Great Lakes basin. Rep. to Great Lakes Sci. Adv. Bd., Int. Joint Comm., Windsor, Ontario. March, 1985.

Sagar, A.D. 1991. Pest control strategies: concerns, issues and options. Environ. Impact Assess. Rev. 11(3):257-79.

Salmon, A. and R.H. Green. 1983. Environmental determinants of unionid clam distribution in the middle Thames River, Ontario. Can. J. Zool. 61:832-838.

Sanders, H.O. 1970. Pesticide toxicity to tadpoles of the western chorus frog (*Pseudacris triseriata*) and Fowler's toad (*Bufo woodhousii fowleri*). Copeia 1970:246-51.

Sanderson, M. 1987. Implications of climatic change for navigation and power generation in the Great Lakes. Environ. Can., Clim. Change Digest CCD 87-03, Toronto.

Schaeffer, J.S. and F.J. Margraf. 1987. Predation of fish eggs by white perch, *Morone americana*, in western Lake Erie. Environ. Biol. Fish. 18(1):77-80.

Schertzer, W. M. and A.M. Sawchuk. 1990. Thermal structure of the lower Great Lakes in a warm year: implications for the occurrence of hypolimnion anoxia. Trans. Am. Fish. Soc. 119(2):195-209.

Schlosser, I.J. 1982. Trophic structure, reproductive success, and growth rate of fishes in a natural and modified headwater stream. Can. J. Fish. and Aquat. Sci. 39:968-78.

Schneberger, E. and J.L. Funk, eds. 1971. Stream channelization: a symposium. Am. Fish. Soc. N. Cent. Div. Spec. Publ. No. 2.

Sedell, J.R. and J.L. Frogatt. 1984. Importance of streamside forests to large rivers: the isolation of the Willamette River, Oregon, USA, from its floodplain by snagging and streamside forest removal. Verh. Int. Ver. Limnol. 22:1828-34.

Sedell, J.R. and F.J. Swanson. 1984. Ecological characteristics of streams in old-growth forests of the Pacific Northwest. pp. 9-16 *in* Fish and wildlife relationships in old-growth forests. Am. Inst. of Fish. Res. Biol. Morehead City, N.C.

Seeb, J.E., L.W. Seeb, D.W. Oates, and F.M. Utter. 1987. Genetic variation and postglacial dispersal of populations of northern pike (*Esox lucius*) in North America. Can. J. Fish. and Aquat. Sci. 44:556-61.

Selgeby, J.H. 1974. Immature insects (Plecoptera, Trichoptera, and Ephemeroptera) collected from deep water in western Lake Superior. J. Fish. Res. Bd. Can. 31:109-111.

Shapiro, J., V. Lamarra, and M. Lynch. 1975. Biomanipulation: an ecosystem approach to lake restoration. *in* P.L Brezonik and J.L Fox, eds. Proceedings of a symposium on water quality management through biological control. Univ. Flor., Gainesville.

Shuter, B.J. and J.R. Post. 1990. Climate, population viability, and the zoogeography of temperate fishes. Trans. Am. Fish. Soc. 119(2):314-36.

Simmons, G.M. and J.R. Voshell. 1978. Pre- and post-impoundment benthic macroinvertebrate communities of the North Anna River. pp. 46-51 *in* J. Cairns, E.F. Benfield, and J.R. Webster, eds. Current perspectives on river-reservoir ecosystems. N. Am. Benth. Soc., Blacksburg, Va.

Simonson, T. and J. Lyons. 1992. Evaluation and monitoring of fish and habitat during priority watershed projects. Wis. Dep. Nat. Resour., Bur. Res., Fish Res. Section Progress Rep. 30 pp.

Smart, M.M., R.G. Rada, D.N. Nielsen, and T.O. Claflin. 1985. The effect of commercial and recreational traffic on the resuspension of sediment in navigation pool 9 of the Upper Mississippi River. Hydrobiologia 126:263-274.

Smith, P. W. 1968. An assessment of changes in fish fauna of two Illinois rivers and its bearing on their future. Trans. Ill. Acad. Sci. 61(1):31-45.

Spangler, G.R., N.R. Payne, J.E. Thorpe, J.M. Byrne, H.A. Regier, and W.J. Christie. 1977. Responses of percids to exploitation. Can. J. Fish. Aquat. Sci. 34:1983-1988.

Spence, J.A. and H.B.N. Hynes. 1971. Differences in benthos upstream and downstream of an impoundment. J. Fish. Res. Bd. Can. 28:35-43.

SPOF. 1983. Strategic planning for Ontario fisheries: the identification of overexploitation. Report of SPOF Working Group No. 15, Ontario Ministry of Nat. Resour. 85 pp.

Staggs, M.D. 1987. Sampling design for fish contaminant monitoring program in Lake Michigan. Wis. Dep. Nat. Resour. Bur. Res. Rep. No. 140.

Staggs, M.D., R.C. Moody, M.J. Hansen, and M.H. Hoff. 1990. Spearing and sport angling for walleye in Wisconsin's ceded territory. Wis. Dep. Nat. Resour. Bur. Fish. Manage. Admin. Rep. No. 31.

Stansbery, D.H. 1970. American malacological union symposium: rare and endangered mollusks. 2. Eastern freshwater mollusks. 1. The Mississippi and St. Lawrence river systems. Malacologia 10(1):9-22.

Stark, W.F. 1988. Wisconsin River of history. [privately printed] 354 pp.

Stewart, D.J., J.F. Kitchell, and L.B. Crowder. 1981. Forage fishes and their salmonid predators in Lake Michigan. Trans. Am. Fish. Soc. 110:751-63.

Stewart, K.W. and C.C. Lindsey. 1983. Postglacial dispersal of lower vertebrates in the Lake Agassiz region. *in* J.T. Teller and L. Clayton, eds. Glacial Lake Agassiz. Geol. Assoc. Can. Spec. Pap. 26:392-419.

Stone, R. 1994. Dioxins dominate Denver gathering of toxicologists. Science 266:1162-1163.

Strahler, A.N. 1957. Quantitative analysis of watershed geomorphology. Trans. Am. Geophys. Union 38:913-20.

Strayer, D.L. and J. Ralley. 1993. Microhabitat use by an assemblage of stream-dwelling unionaceans (Bivalvia), including two rare species of *Alasimidonta*. J. N. Am. Benthol. Soc. 12:247-258.

Sweeney, B.W. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in eastern North America. Proceedings of the Academy of Natural Sciences of Philadelphia 144:291-340.

The Nature Conservancy. 1994. The conservation of biological diversity in the Great Lakes ecosystem: issues and opportunities. TNC Great Lakes Program, Chicago Ill.

Thiel, P.A. 1981. A survey of Unionid mussels in the Upper Mississippi River (pools 3 through 11). Wis. Dep. Nat. Resour., Tech. Bull. 124, Madison. 24 pp.

Threinen, C.W. 1982. The nature of the bait business in Wisconsin: an informational report to the Natural Resources Board. Wis. Dep. Nat. Resour. Adm. Rep. No. 13. 10 pp.

Thuemler, R., A. Hauber, R. Theis, F. Pratt, P. Pajak, M. Staggs, L. Kernen, and D. Gebken. 1989. FERC Project Review Procedures, Vol. 1 Wis. Dep. Nat. Resour. Bur. Fish. Manage., Madison [unpubl. rep.]

Todd, T.N. 1990. Genetic differentiation of walleye stocks in Lake St. Clair and western Lake Erie. U.S. Fish and Wildl. Serv. Fish and Wildl. Tech. Rep. 28.

Tonn, W.M. 1990. Climate change and fish communities: a conceptual framework. Trans. Am. Fish. Soc. 119(2):337-52.

Tonn, W.M. and J.J. Magnuson. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. Ecology 63(4):1149-66.

Treloar, D. and T. Ehlinger. 1991. Walleye stock differentiation and analysis of recruit contributions in the Lake Winnebago system. Wis. Dep. Nat. Resour. Sport Fish Restor. Grant F-83-R Inter. Rep.

Trotzky, H.M. and R.W. Gregory. 1974. The effects of water flow manipulation below a hydroelectric power dam on the bottom fauna of the Upper Kennebec River, Maine. Trans. Am. Fish. Soc. 103:318-324.

Tyus, H.M. 1990. Effects of altered stream flows on fishery resources. Fisheries 15(3):18.

U.S. Department of Energy. 1990. Energy and climate change. Rep. DOE Multi-Lab. Climate Change Comm. Lewis Publishers, Chelsea, Mich. 160 pp.

U.S. Department of the Interior. 1991. Casting light upon the waters: a joint fishery assessment of the Wisconsin ceded territory. U.S. Dep. Int., Bur. Indian Aff., Minneapolis, Minn. 102 pp.

U.S. Department of the Interior/U.S. Department of Commerce. 1993. 1991 National survey of fishing, hunting and wildlife-associated recreation. U.S. Dep. Int., Fish and Wildl. Serv. and U.S. Dep. Commerce, Bur. Census. Washington. 125 pp.

Van Oosten, J. 1945. Turbidity as a factor in the decline of Great Lakes fishes with special reference to Lake Erie. Trans. Am. Fish. Soc. 75:281-322.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Can. J. Fish. and Aquat. Sci. 37:130-37.

Vennum, T. 1988. Wild rice and the Ojibway people. St. Paul Minn Historical Society Press. 358 pp.

Verry, E.S. 1986. Forest harvesting and water: the lake states experience. Water Resour. Bull. Rep. No. 84184:1039-1047.

Vetrano, D.M. 1988. Unit construction of trout habitat improvement structures for Wisconsin coulee streams. Wis. Dep. Nat. Resour. Bur. Fish. Manage. Admin. Rep. No. 27.

Vogt, R.C. 1981. Natural history of amphibians and reptiles of Wisconsin. Milwaukee Pub. Mus., Milwaukee, Wis. 205 pp.

Wall, G. and C. Wright. 1977. The environmental impact of outdoor recreation. Univ. Waterloo, Ontario, Dep. Geography Publication Series 11. 69 pp.

Ward, J.V. 1976. Effects of flow patterns below large dams on stream benthos: a review. pp. 235-253 *in* J.F. Orsborn and C.H. Allman, eds. Instream flow needs symposium, Vol. 2. Am. Fish. Soc., Bethesda.

Watermolen, D.J. 1993*a*. Upper Green Bay basin water quality management plan. Wis. Dep. Nat. Resour. Bur. Water Resour. Manage., Madison.

Watermolen, D.J. 1993*b*. Potential effects of seawall development on amphibian and reptile populations in Lake Winneconne, Winnebago County, Wisconsin - 1993. Wis. Dep. Nat. Resour., Wis. Endang. Res. Rept. (95):1-12.

Waters, T.F. 1983. Replacement of brook trout by brown trout over 15 years in a Minnesota stream: production and abundance. Trans. Am. Fish. Soc. 112:137-146.

Way, C.M., A.C. Miller, and B.S. Payne. 1990. The influence of physical factors on the distribution and abundance of freshwater mussels (Bivalvia:Unionidae) in the lower Tennessee River. Nautilus 103:96-98.

Wells, L. and R.W. Hatch. 1985. Status of bloater chubs, alewives, smelt, slimy sculpins, deepwater sculpins and yellow perch in Lake Michigan, 1984. Pap. presented at the Great Lakes Fishery Commission, Lake Michigan Committee Meeting, Ann Arbor, Mich., 19 March 1985.

Wells, L. and A.L. McLain. 1973. Lake Michigan - man's effect on native fish stocks and other biota. Great Lakes Fish. Comm. Tech. Rep. No. 20.

Williams, D.D. and H.B.N. Hynes. 1977. Benthic community development in a new stream. Can. J. Zool. 55:1071-1076.

Williams, R.D. and R.N. Winget. 1979. Macroinvertebrate response to flow manipulation in the Strawberry River, Utah (U.S.A.). pp. 365-376 *in* J.V. Ward and J.A. Stanford, eds. The ecology of regulated streams. Plenum Press, New York.

Winnell, M.H. and D.S. White. 1985. Ecology of some Chironomidae (Diptera) from southeastern Lake Michigan, U.S.A. Trans. Am. Entomol. Soc. 111:279-359.

Winston, M.R., C.M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. Trans. Am. Fish. Soc. 120:98-105.

Wisconsin Department of Natural Resources. 1980. Wisconsin Trout Streams. Wis. Dep. Nat. Resour. Publ. 6-3600(80).

Wisconsin Department of Natural Resources. 1986. Lake Michigan fisheries management plan. Wis. Dep. Nat. Resour. Admin. Rep. No. 25.

Wisconsin Department of Natural Resources. 1988. Lake Superior fisheries management plan 1988-1998. Wis. Dep. Nat. Resour. Admin. Rep. No. 28.

Wisconsin Department of Natural Resources. 1991. Wisconsin Statewide Comprehensive Outdoor Recreation Plan for 1991-96. Wis. Dep. Nat. Resour.

Wood, J.T. 1949. Observations of (*Natrix septemvittata*) (Say) in southwestern Ohio. Am. Midl. Nat. 42(3):744-50.

Yokley, P., Jr. 1977. The effect of gravel dredging on mussel production. Bull. Amer. Malacol. Un. for 1976: 20-22.

Yokley, P., Jr., and C.H. Gooch. 1976. The effect of gravel dredging on reservoir primary production, invertebrate production and mussel production. Tenn. Wild. Res. Agency, Nashville, (NOAA-76091401. Nat. Tech. Info. Serv. Pb-260 485).

York, D. 1994. Recreational-boating disturbances of natural communities and wildlife: an annotated bibliography. Nat. Biol. Survey Biol. Rep. No. 22, 30 pp.

Yousef, Y.A., W.M. McLellon, and H.H. Zebuth. 1980. Changes in phosphorus concentrations due to mixing by motorboats in shallow lakes. Water Res. 14:841-852.

Zillioux, E.J., D.P. Porcella, and J.M. Benoit. 1993. Mercury cycling and effects in freshwater wetland ecosystems. Environ. Toxicol. Chem. 12:2245-2264.